Pre-fire treatment effects and post-fire forest dynamics

on the Rodeo-Chediski burn area, Arizona

by Barbara A. Strom

A Thesis

Submitted in Partial Fulfillment

Of the Requirements for the Degree of

Master of Science

in Forestry

Northern Arizona University

May 2005

Approved:

Peter Z. Fulé, Ph.D., Chair

Stephen C. Hart, Ph.D.

Thomas E. Kolb, Ph.D.

ABSTRACT

Pre-fire treatment effects and post-fire forest dynamics on the Rodeo-Chediski burn area, Arizona

Barbara A. Strom

The 2002 Rodeo-Chediski fire was the largest wildfire in Arizona history at 189,000 ha (468,000 acres), and exhibited some of the most extreme fire behavior ever seen in the Southwest. Pre-fire fuel reduction treatments of thinning, timber harvesting, and prescribed burning on the White Mountain Apache Tribal lands (WMAT) and thinning on the Apache-Sitgreaves National Forest (A-S) set the stage for a test of the upper boundary of effectiveness of fuel reduction treatments at decreasing burn severity. On the WMAT, we sampled 90 six-hectare study sites two years after the fire, representing 30% of the entire burn area on White Mountain Apache Tribe lands, or 34,000 hectares, and comprising a matrix of three burn severities (low, moderate, or high) and three treatments (cutting and prescribed burning, prescribed burning only, or no treatment). Prescribed burning without cutting was associated with reduced burn severity, but the combination of cutting and prescribed burning had the greatest ameliorative effect. Increasing degree of treatment was associated with an increase in the number of live trees and a decrease in the extremity of fire behavior as indicated by crown base height and bole char height. Ponderosa pine regeneration was very low in untreated areas, with no ponderosa regeneration in high severity untreated areas; over half the study area had no ponderosa regeneration, and 16% of the study area had no ponderosa regeneration and no surviving ponderosa trees. On the A-S, we sampled seven pairs of thinned/unthinned study sites two years after the fire. Thinned areas had more live trees, higher survival, and less extreme fire behavior as indicated by crown base height and bole char height. Ponderosa pine regeneration was patchily distributed over the study sites, and lower in untreated areas. Differences between thinned and untreated areas persisted for several decades after the fire in stand structure characteristics and for at least 100 years in species dominance when modeled using the Forest Vegetation Simulator. Our findings strongly indicated that thinning, timber harvesting, and prescribed burning were associated with reduced burn severity even in an extraordinarily intense fire, provided that the treatments occurred within the decade before the fire. Future forest development on the burn area will most likely take one of two trajectories: recovery to a ponderosa pine/Gambel oak forest or a shift to an alternative stable state such as an oak/manzanita shrubfield, with untreated and high-severity areas more apt to undergo a shift to a shrubfield state.

ACKNOWLEDGEMENTS

For all his support, advice, and amazingly fast turnaround times on all the incarnations this thesis has been through, I thank Pete Fulé. I would like to thank the other members of my graduate committee, Tom Kolb and Steve Hart of the School of Forestry, as well as Carolyn Hull Sieg of the Rocky Mountain Research Station, who was our Forest Service liason for the project. It has been wonderful to work with my colleague on the Rodeo-Chediski project, Amanda Kuenzi.

For research on the White Mountain Apache Tribal lands (Chapter 3), I thank the White Mountain Apache Tribal Council for permission to sample the study sites. Many thanks to Paul Declay, WMAT Forest Manager; Jim Youtz, Ft. Apache Agency BIA Forestry; and Mary Steuver, WMAT Burned Area Emergency Recovery team leader; each provided extensive assistance and cooperation.

For research on the Apache-Sitgreaves National Forest (Chapter 4), I thank the staff of the Apache-Sitgreaves for permission to sample the study sites, and for the maps and other guidance they provided, especially Mike Stefaniak, Gayle Richardson, and Joseph Hamrick; Linda Wadleigh, Southwestern Region Fire Ecologist; and Chad Hoffman, who got us acquainted with the Apache-Sitgreaves sites in 2003, and has been a source of friendship and advice since then.

I am grateful to everyone at the Ecological Restoration Institute for their support and long hours in the field and the lab. Scott Curran was invaluable in getting us set up with electronic data collection, and for his help with database design and data analysis after fieldwork was complete. Joe Crouse got me started with the extensive GIS analysis needed to select the WMAT study sites. Matt Tuten, who was a Rodeo-Chediski crew leader all summer and was a great help in getting ready for the fieldwork season, was vital to the project. For fieldwork over the summer of 2004, I am indebted to many ERI research assistants. Ken Baumgartner – who even came with us on the reconnaissance trip in 2003 – and Shawnte Greenway were with us all summer; Lang Suby, Jeff Rainey, and Chris Gonzales were on the crew for most of the summer, and many thanks to everyone else who worked on the Rodeo-Chediski crew.

This research was funded by a Joint Venture Agreement with the U.S. Forest Service Rocky Mountain Research Station (# 03-JV-11221615-153).

Finally, I thank my friends and family, especially my partner, Janet Hanseth, and my father and mother, David and Edith Strom, for their love and support.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

PREFACE

This thesis contains two manuscript chapters intended for publication. The manuscripts are Chapter 3, "Pre-fire treatment effects and post-fire forest dynamics on the Apache-Sitgreaves National Forest", and Chapter 4, "Pre-fire treatment effects and post-fire forest dynamics on the White Mountain Apache Tribal lands." The text has been edited to minimize redundancy between chapters wherever possible, and Chapter 6, "Literature Cited", includes the literature cited for all chapters.

1. INTRODUCTION

I. Wildfire response to fuel reduction treatments and future forest growth

The 2002 Rodeo-Chediski fire at 189,000 hectares was part of a trend of increasingly larger crownfires occurring in the ponderosa pine forests of the Southwest, which prior to EuroAmerican settlement in the 1870s had a frequent surface fire regime that was then interrupted by overgrazing and fire suppression (Covington and Moore 1994). The scope of fuel reduction treatments such as thinning and prescribed burning intended to reduce crownfire susceptibility has increased recently, but there has not been substantial testing of treatment effectiveness. Only three studies have systematically investigated the effect of fuel reduction treatments on wildfire severity (Martinson and Omi 2003, Cram and Baker 2003). The Rodeo-Chediski fire exhibited some of the most extreme fire behavior ever seen in the Southwest (USFS 2002), and numerous well-documented fuel reduction treatments took place before the fire on both the White Mountain Apache Tribal lands and the Apache-Sitgreaves National Forest, setting the stage for a test of the upper boundary of effectiveness of fuel reduction treatments at reducing burn severity..

II. Research questions

The goals of this research were: 1) to determine the type and frequency of fuel reduction treatments that were effective in reducing burn severity; 2) to characterize the post-fire forest across a range of treatments (and burn severities, for the study on White Mountain Apache Tribal lands);and 3) for the study on the Apache-Sitgreaves National Forest, to project differences in species dominance or conversions in vegetation type across treatments as the burn area recovers.

For the study on the White Mountain Apache Tribal lands, we compared post-fire forest structure and initial recovery across a matrix of three treatments (1. timber harvesting or thinning combined with prescribed burning, 2. prescribed burning only, and 3. no treatment) and three burn severities (low, moderate, or high). Our hypotheses were that in an extraordinarily intense fire (a) stand treatments before the wildfire leads to lower burn severity than in untreated areas; (b) under similar burn severities, post-fire forest conditions still vary across treatments; (c) resistance to crown fire in the near future is greatest in treated and high-severity areas; and (d) post-fire regeneration and shrubs, and thus potentially future forest development, varies across treatments and burn severities; untreated and higher-severity areas are more likely to transition from the pre-fire ponderosa pine forest to an oak-dominated shrubfield.

For the study on the Apache-Sitgreaves National Forest, we compared pre- and post-fire forest structure, initial recovery, and projected future growth using the Forest Vegetation Simulator (Dixon 2003) for seven pairs of thinned/unthinned stands. Our hypotheses were that: (a) pre-wildfire thinning led to lower burn severity than in untreated areas; (b) post-fire tree regeneration and shrubs, and thus potential forest development, differs on thinned and untreated areas, with higher levels of regeneration in untreated areas; (c) differences in post-fire recovery of thinned and untreated areas will persist for several decades following the fire; and (d) untreated areas are more likely to transition from the pre-fire ponderosa pine forest to an oak-dominated shrubfield.

III. Significance

Unless widespread fuel reduction projects are undertaken in the near future, the vast majority of southwestern ponderosa pine forests are likely to experience severe crownfire in the next few decades (Agee and Skinner 2005). Since the Rodeo-Chediski fire burned across many ponderosa pine habitat types (USFS 1987), much of the Southwest may soon resemble the Rodeo-Chediski burn area. Our clarification of the effects of timber harvesting, thinning, and prescribed burning on wildfire severity and post-fire recovery may be applied to ponderosa pine forests throughout the region. Our study has the potential to give land managers insight on how current decisions on timber harvests and fuel reduction treatments will affect forests a century or more into the future, and help them choose alternatives to minimize destruction of forest resources and ensure the sustainability of ponderosa pine forests.

2. LITERATURE REVIEW

I. Fire regime changes in southwestern ponderosa pine forests

Wildfires in the western United States are becoming larger, more frequent, and more intense. The area burned per year in severe crown fires in southwestern ponderosa pine (*Pinus ponderosa*) forests has been steadily increasing, from tens of ha in the 1960s, to tens of thousands of ha in the 1990s, to 189,000 ha (468,000 acres) in the 2002 Rodeo-Chediski fire (Friederici 2003:xvi). This is a significant departure from the previous fire regime of the area. Fire scar studies from across the Southwest indicate that before extensive EuroAmerican settlement began in the 1870s, low-severity surface fires occurred every 2-25 years (Swetnam and Baisan 1996). Traits such as thick, fire-resistant bark, rapid seedling growth, and highly combustible litter suggest that frequent surface fire was part of the evolutionary environment of ponderosa and related long-needled pines (Agee 1998).

Since EuroAmerican settlement, livestock grazing, fire suppression, and selective logging of large trees have vastly changed the ponderosa pine forests of the Southwest. Early EuroAmerican explorers and settlers made note of "open, park-like" stands of mature ponderosa pine (Weaver 1943, Covington and Moore 1994). Livestock grazing had the earliest effect on fire regimes and subsequent forest structure; it reduced the abundance and continuity of grasses that both suppressed most pine regeneration and acted as fine fuels carrying frequent surface fire (Belsky and Blumenthal 1997). Fire suppression began in the late 1800s, and became much more widespread and effective after World War II when air tankers came into use and firefighting operations became more effective (Pyne 1982). Traditional timber operations focused on the removal of the largest, most valuable trees, and precommercial thinning was uncommon until recently (Scurlock and Finch 1997). A number of years with above-average rainfall led to the establishment of large numbers of ponderosa pine trees in the first decades of the 20th century, especially during 1919 (Cooper 1960, Savage et al. 1996).

These combined factors resulted in a forest that today is far denser than it was prior to EuroAmerican settlement. Though Native Americans lived in the Southwest long before extensive EuroAmerican settlement, for the sake of brevity, the term "presettlement" is used to refer to the environment that existed before EuroAmerican settlement. Because of the near-cessation of fire and the slow decay of wood in the arid climate of the Southwest, evidence of trees in the form of stumps, downed logs, and stump holes persists for decades, thus presettlement forest structure can be reconstructed with a good degree of confidence (Covington and Moore 1994, Fulé et al. 1997, Moore et al. 2004). Presettlement forest densities were approx. 70-150 trees/ha, while many current stands are approx. 500-1300 trees/ha; also, average tree size has decreased due to selective logging and the lack of fire-caused thinning (Fulé et al. 1997, Moore et al. 2004). Records of foresters from the early 20th century also confirm that ponderosa forests were once much more open (Lang and Stewart 1910, Woolsey 1911, 1912; Pearson 1923).

These forest structural changes outside the known historic range of variability and resulting fuel buildups are the major factors leading to recent massive wildfires. The density of small-diameter trees has greatly increased, and pine litter has accumulated on the forest floor, thereby increasing canopy fuels, forest floor fuels, and vertical fuel continuity. Climatic fluctuations may also be playing a role in these structural changes, and the current Southwestern drought has certainly been a factor in recent wildfires as well (Covington and Moore 1994, Swetnam et al. 1999).

II. Fuel reduction treatments and their effectiveness

Fuel reduction projects such as thinning of small-diameter trees and prescribed burning in an attempt to decrease the risk of severe crownfire have remained limited in scope until recently (USFS 1999). The Healthy Forests Restoration Act and Healthy Forests Initiative recently increased funding and streamlined the approval process for fuel reduction projects (U.S. Senate 2003). Public reaction has been mixed, particularly to the Healthy Forests Initiative, since the vast majority of current fuel reduction projects have taken place under its categorical exclusions (USDA/USDI 2004). Questions still exist as to what type, frequency, and placement of fuel reduction treatments is the most effective in preventing catastrophic wildfire.

Stand treatments such as timber harvesting including pre-commercial thinning, pre-commercial thinning used alone, prescribed burning, and restoration to presettlement conditions using both mechanical thinning and prescribed burning have been shown to decrease fuel loading and fuel continuity. They do so both by removing many of the small trees that became established at extremely high densities after EuroAmerican settlement, and also by reducing forest floor fuels in the case of prescribed burning. They have been projected to decrease the susceptibility of treated stands to crownfire especially due to their reduction of ladder fuels, which would otherwise allow surface fire to reach the canopy, and their reduction of canopy fuels that would otherwise allow a passive crownfire (torching) to become an active crownfire (Gaines et al. 1958, Deeming 1990, van Wagtendonk 1996, Stephens 1998, Covington et al. 2001, Fulé et al. 2001, Brose and Wade 2002).

While prescribed burning alone reduces fuel loads and may accomplish some thinning, model-based studies have questioned whether prescribed burning alone can be effective in reducing crown fire susceptibility, due to the limited effect of low-intensity fire on canopy and ladder fuels as compared to forest floor fuels (Scott 1998, Fulé et al. 2002). Prescribed burning can lead to high mortality of older trees because of the accumulation of duff around these trees that can expose the cambium to lethal amounts of heat upon combustion, a.k.a. cambial girdling. This mortality can be reduced by raking the duff away from the bases of presettlement trees, but this is labor-intensive and perhaps not always feasible over large areas (Covington and Sackett 1984). The effects and intensity of prescribed burning can be difficult to control; prescribed fires that have gone beyond the prescription in terms of area or behavior have sometimes had severe effects, disinclining land managers from using prescribed fire at a sufficient intensity to accomplish significant thinning (Sackett and Haase 1998). Also, the effects of prescribed fire on forest productivity and wildlife are not well known, leading some to argue for conservative use of fire until additional research has been conducted (Tiedemann et al. 2000).

Several empirical studies have examined adjacent treated and untreated stands after wildfire, reporting that areas that had fuel reduction treatments experienced lower burn severity (Agee et al. 2000); Martinson and Omi (2003) summarized these studies. Several were nonsystematic investigations, focused on slash treatments rather than prescribed burning and thinning, or were based solely on remote-sensing data without field verification (Wagle and Eakle 1979, Omi and Kalabokidis 1991, Vihanek and Ottmar 1993). Weatherspoon and Skinner (1995) found that burn severity, as measured by crown scorch and consumption using aerial photos, was higher in harvested stands, but the stands they sampled had not been pre-commercially thinned prior to harvest. They also found that burn severity was lower in harvested stands where slash had been treated than where it was left untreated. While conventional timber harvesting without pre-commercial thinning does reduce fuels in an absolute sense, it generally does not reduce ladder fuels, which contribute to crownfire susceptibility, so it cannot be strictly considered a fuel reduction treatment (Agee and Skinner 2005).

Numerous studies of fuel treatment effects have taken place on White Mountain Apache Tribal lands. The White Mountain Apache has been documenting harvesting operations since 1937, prescribed burning operations since 1950, and thinning operations since 1979. Gaines et al. (1958) found that a prescribed fire led to significant fuel reduction. Weaver (1955) and Kallander (1969) reported that wildfire severity was minimized on areas with recent prescribed burns compared to untreated areas. Lindenmuth (1962) studied a prescribed fire that was allowed to burn intensely, and reported that more crowning occurred in a recently cut unit due to the presence of untreated slash. After the Rodeo-Chediski fire, Finney et al. (in press) conducted a satellite imagery analysis of fire behavior in prescription burned vs. untreated areas on the Apache-Sitgreaves National Forest and White Mountain Apache Tribal lands, finding that burn severity as assessed by differenced normalized burn ratio was substantially reduced in areas with recent prescribed burns ($\geq$ 9 years old) and that, on their leeward sides, treatments protected nearby untreated areas.

Pollet and Omi (2002) systematically examined the effect of treatments (prescribed burning, whole-tree thinning, or thinning followed by prescribed burning) on burn severity and crown scorch for four wildfires in western ponderosa pine. Omi and Martinson (2002) carried out a similar investigation for four wildfires in ecosystems adapted to frequent fire, including two wildfires in ponderosa pine forests, comparing recent (< 10 years before wildfire) treatments of single and repeated prescribed burning, debris removal, and mechanical thinning with and without slash removal. Both studies found that burn severity and crown scorch were significantly lower in all treated areas. These studies collectively considered treatment effect on wildfires across the West and as such can be used to make a more generalized statement about treatment effect on burn severity. The treatments for each wildfire differed, however, so treatment effect cannot be clearly separated from the effect the location and seasonality of the wildfire may have had on burn severity. Neither study assessed regeneration in order to make predictions about future forest dynamics in treated versus untreated areas, nor did they compare portions of treated and untreated areas with similar burn severities. Pollet and Omi (2002) also questioned whether fuel treatments will have any effect under extreme fire conditions, because drought and high winds may play a more important role in fire behavior than fuels.

Cram and Baker (2003) systematically investigated silvicultural treatment effect (commercial timber, individual tree, and group selection harvests; pre-commercial thinning; forest health or aesthetic treatments; prescribed burning; or combinations of the above) after four wildfires, including the Rodeo-Chediski fire within the Apache-Sitgreaves National Forest. They found that treated areas experienced lower burn severity, ground char, and fireline intensity as estimated from bole char than untreated areas. They sampled treatments that were up to 20 years old, and they did not measure regeneration, or shrubs at the species level. A survey of treated and untreated areas immediately after the Rodeo-Chediski fire on the Apache-Sitgreaves National Forest also indicated that treated areas experienced lower burn severity as indicated by tree survival and crown scorch and consumption (USFS 2002).

III. Recovery of ponderosa pine forests after crownfire, and alternative stable states

Many models exist for projecting forest development, which fall into two general classes: statistical models and process models. Statistical models, which are based on known tree growth patterns and stand dynamics, can be quite accurate over fairly short time periods under environmental conditions similar to those under which the model was developed. Process models, which simulate physiological processes involved in growth and competition, can incorporate environmental change, and may be more appropriate to use for long-term modeling, but require much more complex initiation parameters. Some of these models also predict susceptibility to surface or crown fire, and successional dynamics on a landscape influenced by fire. We used the Forest Vegetation Simulator (FVS), an individual-tree growth and yield statistical model (Dixon 2003). It is initialized with standard mensurational data, and outputs both stand-level and tree-level growth data, which can be analyzed by species. FVS has a variant customized for southwestern ponderosa pine, based on the GENGYM model (Edminster et al. 1991), such that its projected results are in accordance with knowledge of local stand dynamics. FIRESUM (Keane et al. 1989, 1990) is an ecological process model that simulates surface fire effects on forest structure and species composition. It has been used in southwestern ponderosa pine forests (Covington et al. 2001), but its short-term predictive accuracy is most likely lower than that of statistical models. FIRE-BGC (Keane et al. 1996) is a combination of FIRESUM and FOREST-BGC (Running and Coughlan 1998, Running and Gower 1991), a biogeochemical process model; however, because of its complexity and detailed input requirements, FIRE-BGC is not yet validated for use outside the northern Rockies. FVSBGC is another combination of two pre-existing models: FVS and the STAND-BGC process model, which is a modified version of FOREST-BGC. FVSBGC provides an interface between STAND-BGC and FVS such that FVS provides the input data for STAND-BGC, which in turn can provide data to FVS. The combination allows modeling of fires as well as forest response to different climate and CO_2 emissions scenarios (Milner et al. 2003). We had originally intended to use FVSBGC to project future growth on the Rodeo-Chediski burn area over a range of climate change scenarios; however, test runs indicated that the model requires further calibration before it can be used in southwestern ponderosa pine.

There has been extensive empirical research on the recovery of pine-dominated forests after wildfire (Foxx 1996, Barton 2002, Gracia et al. 2002, Greene et al. 2004, Savage and Mast 2005). Several studies have also modeled forest dynamics after wildfire, though not necessarily in pine forests (He et al. 2002, Retana et al. 2002, Chapin et al. 2003). A number of these studies have questioned whether forests that historically had a frequent fire regime are resilient to crownfire.

There is widespread concern about future development of arid forests such as those of the Southwest as large, severe crownfires continue (Hessburg et al. 2005). Studies in Arizona (Barton 2002), Mexico (Fulé et al. 2000), and Spain (Retana et al. 2002) have indicated that intense fire in pine-oak forests may result in a shift to a more oak-dominated forest or a conversion to an alternative steady state such as a shrubfield. These shifts in species dominance and conversions in vegetation type to what appears to be an alternative stable state – i.e., from a ponderosa pine forest to an oak/manzanita shrubfield – have been documented after crownfire in several dense southwestern ponderosa pine forests by Savage and Mast (2005). Alternative stable states are self-perpetuating species assemblages that are distinct from the typical assemblage found in a given environment. Relative to the stature and biomass of the historical plant structure, alternative states may be shorter in height (e.g., shrubs vs. trees), with less total biomass, and appear to be an earlier sere, though without evidence of near-term shift back toward the pre-crownfire forest.

A shift to such an alternative state can have major consequences for ecosystem functions and potential land uses (Laycock 1991, Petraitis and Latham 1999). For instance, Gambel oak (*Quercus gambelii*) is the major oak species on the Rodeo-Chediski burn area; while ponderosa pine is the most commercially valuable timber species in the Southwest, the only common wood product uses of Gambel oak are fuelwood and occasionally fenceposts (Barger and Ffolliott 1972, Johnson 1985, Blatner and Govett 1988). Some wildlife species, such as Abert and Kaibab squirrels (*Sciurus aberti* and *S.*

aberti kaibabensis, respectively), are dependent on ponderosa pine (Keith 1965, Linhart 1988).

IV. Research approach

No post-wildfire research has been done on a large burn area investigating the effect of multiple treatment types on forest survival and initial regrowth across a range of burn severities, and little research has been done investigating the effect of fuel reduction treatments both on burn severity and projected future forest dynamics. Because of its extreme fire behavior (USFS 2002) and large size, the Rodeo-Chediski fire can serve as a test case of the upper boundary of response to fuel treatments in a southwestern forest. The goals of this research were to determine the type and frequency of fuel reduction treatments that were effective in reducing burn severity, to characterize the post-fire forest across a range of treatments and burn severities, and to project differences in species dominance or conversions in vegetation type across treatments as the burn area recovers.

3. PRE-FIRE TREATMENT EFFECTS AND POST-FIRE FOREST DYNAMICS ON WHITE MOUNTAIN APACHE TRIBE LANDS WITHIN THE RODEO-CHEDISKI BURN AREA, ARIZONA

Abstract

The 2002 Rodeo-Chediski fire was the largest wildfire in Arizona history, and exhibited some of the most extreme fire behavior ever seen in the Southwestern United States. On the White Mountain Apache Tribal lands within the burn area, pre-fire fuel reduction treatments of thinning, timber harvesting, and prescribed burning set the stage for a test of the upper boundary of effectiveness of fuel reduction treatments at decreasing burn severity. We sampled 90 six-hectare study sites two years after the fire, representing 30% (34,000 ha) of the entire burn area on White Mountain Apache Tribe lands, and comprising a matrix of three burn severities (low, moderate, or high) and three treatments (cutting and prescribed burning, prescribed burning only, or no treatment). Our findings indicate that thinning, timber harvesting, and prescribed burning were associated with reduced burn severity even in an extraordinarily intense fire, provided that the treatments occurred within the decade before the fire. Prescribed burning without cutting reduced burn severity considerably, but the combination of cutting and prescribed burning had the greatest ameliorative effect. While burn severity explained more of the variation in forest structure than did treatments, increasing degree of treatment was associated with an increase in the number of live trees and a decrease in the extremity of fire behavior as indicated by crown base height and bole char height. Ponderosa pine regeneration was very low in untreated areas, with no ponderosa regeneration whatsoever in high severity untreated areas. Over half the study area had no ponderosa regeneration,

and 16% of the study area had no ponderosa regeneration and no surviving ponderosa trees. Future forest development will most likely take one of two trajectories: recovery to a ponderosa pine/Gambel oak forest or type conversion to an oak-dominated shrubfield, with untreated and high-severity areas more apt to undergo a type conversion.

Introduction

Wildfires in frequent fire-adapted ecosystems of the western U.S. are becoming larger, more frequent, and more severe largely because of changes in forest structure caused by fire suppression, selective logging of large trees, and livestock overgrazing occurring after widespread EuroAmerican settlement (Covington and Moore 1994, Swetnam et al. 1999). Stand treatments such as timber harvesting, thinning, and prescribed burning have been shown to decrease fuel loads and have been projected to decrease the susceptibility of treated stands to crown fire (Deeming 1990, van Wagtendonk 1996, Stephens 1998, Covington et al. 2001, Fulé et al. 2001). While prescribed burning alone reduces fuel loads and may accomplish thinning, model-based studies have questioned whether prescribed burning alone can be effective in reducing crown fire susceptibility, due to the limited effect of low-intensity fire on canopy and ladder fuels as compared to forest floor fuels (Scott 1998, Fulé et al. 2002). Prescribed burning can also lead to high mortality of older trees, and its effects can be difficult to control (Sackett and Haase 1998).

Several empirical studies have examined adjacent treated and untreated stands after wildfire, reporting that areas that had fuel reduction treatments experienced lower burn severity (Agee et al. 2000, Martinson and Omi 2003). Several were nonsystematic investigations, focused on slash treatments rather than prescribed burning and thinning, or were based solely on remote-sensing data without field verification (Wagle and Eakle 1979, Omi and Kalabokidis 1991, Vihanek and Ottmar 1993, Weatherspoon and Skinner 1995). Pollet and Omi (2002) and Omi and Martinson (2002) systematically examined the effect of fuel reduction treatments on burn severity and crown scorch for a total of six wildfires in western ponderosa pine. Both studies found that burn severity and crown scorch were significantly lower in all treated areas. These studies collectively considered treatment effect on wildfires across the West and as such can be used to make a more generalized statement about treatment effect on burn severity. The treatments for each wildfire differed, however, so treatment effect cannot be clearly separated from the effect the location and seasonality of the wildfire may have had on burn severity. They also did not assess potential regrowth by measuring regeneration or shrubs. Cram and Baker (2003) systematically investigated silvicultural treatment effect after four wildfires, including the Rodeo-Chediski burn area within the Apache-Sitgreaves National Forest. They found that treated areas experienced lower burn severity, ground char, and fireline intensity as estimated from bole char. They sampled treatments that were up to 20 years old and did not measure regeneration or shrubs at the species level. No post-wildfire research has been done on a large burn area investigating the effect of multiple treatment types on forest survival and initial regrowth across a range of burn severities.

The Rodeo-Chediski fire was the largest wildfire in Arizona's recorded history (Friederici 2003:xvi). It burned approximately 189,000 ha (468,000 acres) from June 18 - July 7, 2002, largely on the White Mountain Apache Tribe lands and the Apache-

Sitgreaves National Forest (USFS 2002), leaving a mixed pattern of burn severity (Figure 1). Overall fire behavior was extreme: this was a plume-dominated wind, fuel, and topographically driven fire, very different from previous wind-driven large fires which tended to assume a long, narrow shape pointing to the northeast. Plume collapses occurred up to five times per day, flame lengths on both the White Mountain Apache Tribe lands and the Apache-Sitgreaves National Forest reached 60-120 m (200-400 ft), and the rate of spread was 6.4 km/hour (4 mph) at its maximum (USFS 2002). Fuel moisture content for all size classes was below 4% for the duration of the fire, and afternoon wind gusts exceeded 40 km/h (25 mph) on most days and 73 km/h on June 21 (45 mph; Finney et al., in press).

If current wildfire trends continue, the vast majority of southwestern ponderosa pine forests are likely to experience severe crownfire in the next few decades (Agee and Skinner 2005). Temperature and precipitation are expected to increase in the Southwest due to climate change, which may also increase the frequency and total area burned by severe fires (McKenzie et al. 2004). Much of the Southwest may resemble the Rodeo-Chediski burn area; thus, determining the potential future trajectories of the burn area becomes even more important. Three studies of forest regrowth after wildfire, in Arizona (Barton 2002), Mexico (Fulé et al. 2000), and Spain (Retana et al. 2002) have projected that intense fire in pine-oak forests may result in a conversion to an oak-dominated forest or shrubfield. This conversion to a shrubfield state has been confirmed to persist up to five decades after some crownfires in dense ponderosa pine forests by Savage and Mast (2005). Previous studies have questioned whether fuel treatments will have any effect under extreme fire conditions (Pollet and Omi 2002). The Rodeo-Chediski served as a test case of the upper boundary of response to fuel treatments.

Our hypotheses were that in an extraordinarily intense fire: (a) stand treatments (prescribed burning alone, and timber harvesting or thinning combined with prescribed burning) before the wildfire led to lower burn severity than in untreated areas; (b) under similar burn severities, post-fire forest conditions still vary across treatments; (c) resistance to crown fire in the near future is greatest in treated and high-severity areas; and (d) post-fire regeneration and shrubs, and thus potentially future forest development, varies across treatments and burn severities; untreated and higher-severity areas are more likely to transition from the pre-fire ponderosa pine forest to an oak-dominated shrubfield.

Methods

Study Area

The Rodeo-Chediski burn area's 189,000 hectares (468,000 acres) span the Mogollon Rim in east-central Arizona. The majority of the burn area is within the northwestern portion of the White Mountain Apache Tribe lands (WMAT) to the south of the Mogollon Rim and the Apache-Sitgreaves National Forest to the north of the Mogollon Rim. This study focused on the portion of the burn area on the White Mountain Apache Tribe lands. July maximum air temperature is 29.2ºC (84.5ºF), January minimum temperature is -7.8 ºC (17.9 ºF), annual precipitation is 50.6 cm (19.9 in), and annual snowfall is 99.3 cm (39.1 in); these are 1971-2000 averages (excepting snowfall, a 1950-2004 average) from the Heber Ranger Station on the northwestern edge of the burn area. The study sites ranged in elevation from 2,000 – 2,295 m (6,560 – 7,530 ft). Forests on mid-elevation sites were dominated by ponderosa pine (*Pinus ponderosa*) with Gambel oak (*Quercus gambelii*), alligator juniper (*Juniperus deppeana*), New Mexico locust (*Robinia neomexicana*), and pinemat manzanita (*Arctostaphylos pungens*). Higher-elevation sites included ponderosa pine, white fir (*Abies concolor*), and Douglas fir (*Pseudotsuga menziesii*). Soils are largely of the Overgaard Series, with some from the Haplustolls and Elledge Series; soil types ranged from gravelly loam and gravelly fine sandy loam to rock outcrop complexes. The topography was rugged in comparison to the portion of the burn area on the Apache-Sitgreaves National Forest north of the Mogollon Rim.

The Rodeo-Chediski fire presents an excellent opportunity for a natural experiment on the effectiveness of fuel treatments at reducing burn severity. Finney et al. (in press) conducted a GIS analysis of fire behavior and treatment areas on the Rodeo-Chediski burn area, finding that burn severity as assessed by differenced normalized burn ratio was substantially reduced in treated areas; their results have not yet been verified in the field, however. The WMAT has been documenting their harvesting operations since at least 1937, prescribed burning operations since at least 1950 (Weaver 1951) and thinning operations since at least 1979, so detailed spatial data is available on fuel reduction treatments. Perhaps due to these meticulous records, some of the earliest comparisons of wildfire effect on treated vs. untreated areas took place on WMAT lands (Weaver 1955, Kallander 1969). Because the fire moved so quickly on WMAT lands that extensive suppression operations could not be undertaken safely, the effect of backfires, burnouts, and other suppression operations on burn severity versus uncontrolled wildfire behavior was minimized (USFS 2002). While we did not have fine-scale data on fire behavior – i.e., whether a given stand burned during the day or night, or during a plume collapse – and thus could not determine the effect of transient environmental conditions on burn severity, a large number of widely distributed study sites should mitigate their cumulative effects.

Study Site Selection

We gauged burn severity according to a differenced normalized burn ratio (ΔNBR) map created by the National Park Service/US Geological Survey Burn Severity Mapping Project (2002). ΔNBR is calculated from Landsat ETM+ images taken before and after the fire using the ratio of near-infrared (band 4) to mid-infrared (band 7) reflectance. ΔNBR has been shown to correspond well with burn severity in the Southwest (Miller and Yool 2002, van Wagtendonk et al. 2004). No ΔNBR severity thresholds have been field-verified on WMAT lands, so our severity classification thresholds were modified from of those developed for the 2001 Leroux fire in the Coconino National Forest, Arizona, approximately 150 km from the Rodeo-Chediski burn area (Cocke et al., in press); we combined the unburned and low severity categories because of the very low percentage of unburned area within the fire perimeter. The burn severity map of the Rodeo-Chediski fire generated using these thresholds (low: < 240; moderate: 241-570; high: > 570) is shown in Figure 3.1.

The hypothesis of treatment effect on burn severity was inherent to our sampling design, especially with regard to the treatment period; this allowed us to investigate the effect of different treatments on areas with similar burn severities. We compared the burn severity distribution of prescribed burning treatments with different years as starting points of the treatment period (Figure 3.2), using a two-sample Kolmogorov-Smirnov test of a 0.5% subsample of the raw ΔNBR data. We chose to sample treatments that took place within the ten years before the fire (1991-2001), because the burn severity distribution for treatment periods starting before 1991 was not significantly different from that of the entire burn area. The 10-year time period was also recognized in the Forest Service fire summary report as the most effective (USFS 2002). While prescribed burning was associated with decreased burn severity, the combination of cutting and prescribed burning had the greatest ameliorative effect (Figure 3.3). Pairwise comparisons of the burn severity distributions for each treatment type were significantly different, according to the same Kolmogorov-Smirnov method that was used to determine the longest effective prescribed burning treatment period.

We sampled only the treatments of cut and prescribed burn, prescribed burn only, and no treatment. "Cutting" included uneven-aged forest management and non-commercial thinning followed by slash disposal and broadcast burning. Uneven-aged management guidelines followed a range of a Q-slope factor of 1.1 – 1.3, an SDI maximum of 1,110 25.5-cm trees/hectare (450 10-inch trees/acre), and maximum diameter of 46-76 cm (18-30 in) (Youtz 2003). Out of 111,837 hectares (276,355 acres) burned on WMAT land by the R-C fire, 13% were cut and prescription burned and 9.4% were only prescription burned during or after 1991 (Table 3.1).

Ninety 6-hectare study sites were sampled in May-August of 2004, comprising a matrix of three burn severities (low, moderate, or high) by three treatments (cutting and prescribed burning, prescribed burning only, or no treatment). In order to obtain a more accurate spatial representation of all treatment/severity combinations, study sites were distributed equally across treatments and burn severities, with approximately ten sites per treatment/severity combination. Study sites were situated on areas at least 2000 m in elevation and no greater than 45% slope, based on 10-meter resolution USGS DEMs obtained from GeoComm International Corp. (www.geocomm.com). We used the elevational constraint to ensure that our study sites would be in forested areas dominated by ponderosa pine, since some lower-elevation portions of the burn area are piñon-juniper woodlands. The slope constraint ensured that we would not be comparing treated areas on relatively gentle slopes to areas that could not have feasibly been treated because they were on unworkably steep slopes.

GIS coverages of treatment history included spatial data on thinning operations beginning in 1979, prescribed burning operations beginning in 1950, and timber harvesting operations going back to 1937. Over the entire treatment history (1937-2001) of the sample area, and grouping all treatment types, the “cut and burn” area had a total of 4.8±1.7 treatments, the “burn only” area had 2.9±1.2 treatments, and the “no treatment” area had 2.8±1.7 treatments (occurring prior to 1991, in the case of the “no treatment” area). The “cut and burn” area appears to have been more intensively managed than the “burn only” area, given that they differ by more than a single “cut” treatment occurring between 1991 and 2001. The sample area within elevation and slope constraints was 30% of the entire burn area on White Mountain Apache Tribe lands, or 34,000 hectares. Thirty-two percent of this sample area was cut and prescription burned and 15% only prescription burned during or after 1991 (Table 3.1).

Study site locations were randomly chosen within the treatment/severity combinations using GIS. Study sites and the area sampled within treatment/severity categories and elevation/slope constraints are shown in Figure 4. A systematic grid of five plots was established on each study site, for an estimated total of 90 x 5 = 450 plots. Our final study site distribution was not fully balanced. Ten sites were measured in most treatment/severity combinations, but eight sites were measured in burn only/low severity areas, and twelve in burn only/moderate severity areas. Ten sites were measured on no treatment/high severity and cut and burn/high severity areas, but 47 rather than 50 plots were sampled on each of these areas. A total of 444 plots were sampled across all treatment/severity combinations. Plots were located using a Garmin GPS 12 with ten-meter resolution. If the original location of a plot fell on a slope of greater than 45%, the plot center was moved to the nearest area where the majority of the plot would be less than 45% slope; original plot locations that would have included roads were similarly moved.

Measurements

Overstory trees were measured on a variable-radius plot using a prism with a BAF of approximately 2.2 m^2/hectare per tree (10 ft^2/acre per tree). Tree measurements included: tree species, condition, diameter at breast height (~1.4m), total height, canopy base height, bole char height (minimum and maximum), and dwarf mistletoe rating (Hawksworth 1977). Tree condition classes followed Thomas' (1979) description and included: live, declining, and four stages of snags (recent snag, loose-bark snag, clean snag, and snag broken above breast height). Since field measurements were taken two years after the fire, we did not attempt to estimate foliage scorch, as the majority of scorched needles had already fallen from the trees. A subsample of trees (the live specimens of the first four trees on the plot; trees were numbered starting at north and proceeding clockwise around the plot) were cored in order to produce age and growth increment data. Tree increment cores were surfaced and crossdated (Stokes and Smiley, 1968) following standard methods or rings were counted for cores that could not be crossdated, e.g., some junipers. For cores that missed the pith, additional years to the center were estimated with a pith locator (Applequist 1958).

Tree regeneration (saplings and seedlings below breast height) and shrubs were measured on a 40.5 m^2, 3.6 m radius plot (1/100 acre) with origin at plot center. This regeneration/shrub subplot was intended to correspond to the CFI (Continuous Forest Inventory) Minor Plot 2 described by Vandendriesche (1994) for ease of comparison with the CFI plots on the White Mountain Apache Tribal lands. Tree regeneration and shrubs were tallied by species, condition (living or dead), and height class (0-40 cm, 41-80 cm, 81-137 cm, or exact height if > 137 cm). We measured forest floor fuels on a 15.24-meter (50-foot) planar transect at a random azimuth from each plot center, using Brown's (1974) method. Coefficients for planar transect calculations are from Sackett (1980). Coefficients used to convert litter and duff depth to forest floor fuel loadings in megagrams/hectare are from Ffolliott et al.'s (1968) measurements of forest floor weight in northern Arizona ponderosa pine stands.

We took two photographs of each plot: a hemispherical photo using a digital camera with a 180º fisheye lens (Nikon CoolPix E4300 using FC-E8 Fisheye Converter

Lens with UR-E4 Converter Adapter) at plot center in order to record canopy cover, and a plot photograph using a standard lens (Canon PowerShot A70) from 12 m east of plot center. Hemispherical photos were analyzed with Gap Light Analyzer (Institute of Ecosystem Studies 1999) in order to determine forest canopy structure and gap light transmission indices.

Data Analysis

While our data were parametric, inspection of the measurement distributions using frequency histograms, the Shapiro-Wilkes W statistic, and Levene's test revealed that only a few measurements were both normally distributed and had equal variances. We used DISTLM (Anderson 2004), which performs a distribution-free, distance-based multifactor multivariate analysis of variance using permutation. We carried out 999 permutations for each test, and used the Bray-Curtis dissimilarity distance measure for each test. The alpha used to denote a significant difference was 0.05. Since the method used by DISTLM calculates an exact *p*-value, it has been argued that it is not subject to alpha inflation (Anderson 2001, Anderson and Robinson 2001). When testing overall regeneration and shrubs, we performed both an unstandardized test and a test with the abundance of each species standardized by its total abundance across all plots, in case the dominant species were skewing the results.

Results

Fuel reduction treatments within the decade before the fire were consistently associated with a large reduction in burn severity as measured by remote sensing using ΔNBR (Figure 3.3). Prescribed burning that occurred more than eleven years before the Rodeo-Chediski fire did not appreciably decrease burn severity (Figure 2). Sites that had prescribed burns had considerably reduced severity, but the combination of cutting and prescribed burning was associated with the largest reduction in burn severity.

Ponderosa pine (*Pinus ponderosa*) and Fendler's ceanothus (*Ceanothus fendleri*) were the dominant species in the burn area. Alligator juniper (*Juniperus deppeana*), Gambel oak (*Quercus gambelii*), and Douglas-fir (*Pseudotsuga menziesii*) were also common in the overstory, after which abundances of other species dropped sharply (Table 2). Fendler's ceanothus was by far the most abundant shrub, though pinemat manzanita (*Arctostaphylos pungens*) was also present on many sites (Table 3.3).

Stand-level measurements show that fire effects associated with extreme fire behavior were reduced in treated areas. Forest structure attributes, including density, basal area, crown base and bole char height, and diameter distribution, were all significantly related to both degree of treatment and severity levels, and generally showed a trend with increasing degree of treatment, i.e., from no treatment to burn only to cut and burned. These forest structure attributes displayed much more variation in response to burn severity than to treatments – though differences between treated and untreated areas became more substantial as burn severity increased – and did not exhibit an interaction between treatment and severity (Table 3.4).

Tree density was highest in the burn only treatment, and decreased sharply with increasing severity in untreated areas (Figure 3.5). There was a shallower decline in density with increasing severity in treated areas. Pairwise comparisons indicate that density differed among each of the treatments, across severities (Table 3.4). Approximately 12% of the study area had no surviving trees.

Basal area significantly declined with increasing burn severity, similar to the trend in tree density, and increased with degree of treatment in moderate and high severity areas (Figure 3.6). Basal area in untreated vs. cut and burned areas was significantly different, but prescribed burning did not lead to a significantly different basal area from untreated areas or from cut and burned areas.

Crown base height generally increased with increasing burn severity and degree of treatment in treated areas. Bole char height also increased with burn severity, but decreased with degree of treatment, indicating more extreme fire behavior in untreated areas (Figure 3.7). The progression in crown base height with increasing burn severity was not seen in untreated areas. While crown base height in burn only areas is not significantly different from that of untreated areas ($p = 0.86$) and is different from that of cut and burned areas ($p = 0.001$), bole char height for all trees in burn only areas is not significantly different from that of cut and burned areas ($p = 0.251$) and is different from that of untreated areas ($p = 0.008$).

The diameter distribution for ponderosa pine was particularly indicative of the difference between treated and untreated areas; very few small trees were left on moderate and high severity untreated areas (Figure 3.8). The archetypal reverse-J distribution was present in low severity areas, albeit in a modified form, as the distribution peaked at 10-20 cm trees rather than the smallest size class. Moderate severity portions of the burn only treatment also exhibited a reverse-J distribution with much greater amplitude than that of moderate severity portions of the cut and burn treatment, possibly due to the pre-fire thinning on the cut and burn treatment; these treatments were significantly different.

The vast majority of trees, regardless of treatment or severity, were less than 100 years old (Figure 3.9). The cumulative age frequencies did not display a clear trend with regard to treatment or severity, but treated areas appeared to have proportionally more old trees, especially as burn severity increased.

Snag density generally increased with increasing burn severity and decreased significantly with degree of treatment (Figure 3.10). There were 5-50 snags per hectare > 30 cm DBH, and 1-5 snags per hectare > 50 cm DBH.

Initial post-fire recovery as indicated by fuel loadings, regeneration, and shrubs was significantly related to treatment and burn severity levels for all characteristics measured except densities of a few individual species. However, there were virtually no apparent trends in response to degree of treatment or increasing severity, as there were for most aspects of post-fire forest structure, and there were significant interactions between treatment and severity for nearly all aspects of fuels, regeneration, and shrubs (Table 3.4).

Fuel loadings decreased with increasing burn severity, though this trend was mostly seen in the forest floor fuels; there was a significant treatment/severity interaction effect (Figure 3.11). Fuel loadings were significantly higher in untreated areas than in cut and burned areas. Fine woody fuels (1-100H) and to some extent coarse woody fuels

(1000H) were similar across treatments and severities. The higher amount of coarse woody fuels in untreated high severity areas may have been due to a larger number of trees already having fallen since the fire.

Regeneration was dominated by sprouting species such as Gambel oak and other oaks, and alligator juniper, as well as by New Mexico locust (Figure 3.12). Ninety-six percent of our study area had regeneration of Gambel oak, 97% had Gambel oak or another oak species, 52% had New Mexico locust, and 53% had alligator juniper. While regeneration was significantly linked to treatment and severity, a strong interaction effect prevailed over any trends with regard to degree of treatment and severity levels. The unstandardized test and the test standardized by species had very similar results, the only exception being that the unstandardized test showed that burn only and untreated areas were not significantly different, while the standardized test showed significant pairwise differences among all treatments. Considered separately, neither Gambel oak nor New Mexico locust showed a significant treatment effect either overall or in any of the pairwise comparisons, though New Mexico locust did display an interaction effect (Table 3.4).

Ponderosa pine regeneration was ample in moderate severity treated areas, but there was little regeneration in untreated areas, and no regeneration was observed in untreated areas of high severity (Figure 3.13). In treated areas, ponderosa regeneration increased greatly from low to moderate severity areas, with high severity areas having an intermediate amount. There was no significant severity effect when all treatments are considered, though there was an interaction effect of treatment and severity and an apparent trend in treated areas; all pairwise treatment comparisons were significantly different. Some regeneration within the taller height classes may have survived the fire, especially in low severity areas of the cut and burn treatment. Over all treatments and severities, 57% of our study sites had ponderosa regeneration; broken down by treatment, this was 20% of untreated sites, 66% of burn only sites, and 83% of cut and burned sites. Expanding this up to the landscape level, untreated high severity areas, which had no ponderosa regeneration, constituted 12% of the study area. Taking all treatments into account, approximately 54% of the study area had no ponderosa pine regeneration, and 16% of the study area had no ponderosa pine regeneration and no surviving ponderosa pine trees.

Shrubs on the burn area were overwhelmingly dominated by Fendler's ceanothus, which responded strongly to increasing burn severity and was most abundant in the burn only treatment (Figure 3.14). Overall shrubs (standardized by abundance of each species and unstandardized) showed significant effects for treatment, severity, interaction, and all pairwise treatment comparisons. The significant treatment effect for Fendler's ceanothus was largely due to its abundance in the burn only treatment, as there was no significant difference in its abundance between cut and burned and untreated areas. Pinemat manzanita and other manzanita species were also common. Unlike Fendler's ceanothus and overall shrubs, manzanita density did not differ significantly in burn only and untreated areas, and there was no interaction effect.

Discussion

The Rodeo-Chediski fire exhibited some of the most extreme fire behavior ever seen in the Southwest (USFS 2002), but its behavior and intensity still responded to pre-fire fuel reduction treatments. These treatments led to significant variations in forest structure and initial post-fire recovery even under similar burn severities over the landscape matrix, and will most likely affect future forest dynamics on the burn area.

Sampling two years post-fire offers advantages with regard to estimating future forest development. Because it can take several years for trees to die post-wildfire, we can be more certain which trees have survived the fire; McHugh and Kolb (2003) established that most post-fire mortality of ponderosa pine in northern Arizona occurs by two years after the wildfire. With the passage of two growing seasons since the fire, regeneration was well-established, so we have a better estimate of potential future forest development.

We were not able to take a complete sample of dead trees, and so did not try to reconstruct pre-wildfire forest structure. Dead trees were salvaged from the burn area before we took our field measurements. We included the dead trees that were measured only when calculating bole char height and snag availability for wildlife.

Because we only sampled one burn area and did not treat the Rodeo-Chediski fire complex as two separate fires, our study sites are pseudoreplicates, as is common in studies of wildfire effects (van Mantgem et al. 2001). This limits the causal inference that can be made from statistically significant differences among treatments and burn severities. Given that suppression effects were limited and that the area sampled was restricted in its slope and elevational range, it does make sense to attribute clear trends seen especially in post-fire forest structure with increasing degree of treatment or burn severity to the direct effects of treatment and burn severity.

Pre-wildfire treatment effects

Recent fuel reduction treatments were consistently associated with a substantial reduction in burn severity, based on remotely-sensed data. This result was in general agreement with Finney et al.'s (in press) results for the Rodeo-Chediski fire. While the combination of cutting and prescribed burning had the most ameliorative effect, prescribed burning alone also considerably reduced burn severity. Finney et al. (in press) did not include harvesting or thinning in their comparison of treated and untreated areas, however, instead combining cut and burned and burn only areas. This may have led them to overestimate the effectiveness of prescribed burning. For greatest efficacy, we found that prescribed burning treatments must have taken place within the eleven years before the fire. This is within the 10- to15-year range of effectiveness estimated by Agee and Skinner (2005), and slightly longer than the 9-year period reported by Finney et al. (in press). Finney et al. also used a different burn severity classification, which included an unburned category; this may explain some of the dissimilarity in our results.

Our stand-level measurements also showed that fire behavior and effects were ameliorated in treated areas, in accordance with Cram and Baker's (2003) findings on the Rodeo-Chediski burn area, and with other studies of wildfire effects in treated vs. untreated areas (Pollet and Omi 2002, Omi and Martinson 2002). Treatment effects on post-fire forest structure increased, rather than diminished, as burn severity increased. There were more live trees in treated areas, in both density and basal area. Untreated moderate severity areas had 90% fewer trees than untreated low severity areas, as compared to a 66% difference between low and moderate severity cut and burned areas, exemplifying the increasing effect of treatment as burn severity increased. The abundance of live trees in the smaller size classes was also greater in treated areas as burn severity increased. Although prescribed burning did not significantly change crown base height compared to untreated areas, it appears to have decreased tree density and fuel loadings enough to make stands fairly resistant to crownfire, as exemplified by the similarity of bole char height in the burn only and cut and burn treatments. The fire was most likely responsible for the increased crown base height in moderate severity areas, and may have killed trees rather than merely increasing their crown base height in high severity areas. Crown base height may have been highest in the cut and burn treatment because of a higher initial crown base due to thinning. Increasing degree of treatment was significantly associated with an increase in the number of live trees and crown base height, and a decrease in bole char height.

Future forest development

Regeneration two years after the fire is very different from the current and pre-fire overstory; oaks are present on nearly the entire study area, and alligator juniper and New Mexico locust each occur on about half the study area.

The very low regeneration rate of ponderosa pine on untreated areas, especially those of high severity, is a particular cause for concern due to the large proportion of untreated areas in relation to the study area. Our results imply that there was virtually no pine regeneration on over half of the study area. Especially worrisome is that 16% of the study area had neither ponderosa regeneration nor surviving adult ponderosas to act as seed sources. Treated areas had much more pine regeneration than untreated areas, in terms of both density and percent of sites on which regeneration occurred.

The abundance of Fendler's ceanothus in high-severity areas and the burn only treatment, reaching densities in the tens of thousands per hectare, may compete with regeneration, thus slowing recovery of the overstory. However, ceanothus also provides excellent wildlife forage, and as a nitrogen fixer, may hasten soil recovery. Manzanita may have constituted a similar percent cover as Fendler's ceanothus despite its low density due to its much larger size, and so may also be inhibiting tree regeneration in some areas.

Future growth will most likely be a more evenly balanced mixture of Gambel oak and other oaks, juniper, and ponderosa pine, rather than the strongly ponderosa-dominated forest that existed before the fire. New Mexico locust will also play a large role for at least a few decades. Savage and Mast (2005) delineated several trajectories that extremely dense southwestern ponderosa pine forests have taken after crown fire. Five of the ten burn areas they studied either overlapped or were within approx. 100 km of the Rodeo-Chediski burn area. These post-crownfire trajectories include: a return to a dense pine forest; a conversion in vegetation type to an alternative state, i.e., an oak shrubfield or grassland; or a transition to a ponderosa pine/Gambel oak early successional forest. They found that these alternative states had persisted for at least fifty years after fire. Moir and Dieterich (1988) proposed a comparable alternative end result of succession in ponderosa pine ecosystems, suggesting that in addition to open old-growth forests, meadows may also be perpetuated by frequent low-intensity fire. Since oaks were present on nearly the entire study area, a conversion to grassland is doubtful. Given that half the study area does not yet have ponderosa pine regeneration, a type conversion away from a ponderosa pine-dominated forest to a shrubfield with oak, locust, manzanita, and some juniper is likely, especially on the 16% of the landscape where there was no surviving ponderosa pine and no ponderosa regeneration. On the highest-severity areas, recovery even to this shrubfield type will be slow. Cut and burned areas and the majority of the burn only treatment may more quickly recover to a ponderosa pine/Gambel oak forest similar to pre-fire conditions, with the balance shifted more in favor of oaks.

Crownfire resistance in the near future appears to still be highest in the cut and burn treatment, and also in higher-severity areas of all treatments, based on overall tree density and density of small trees, forest floor fuels, and crown base height. Tree density, which is indicative of horizontal fuel loadings and continuity, was lowest in the cut and burn treatment, high severity burn-only areas, and moderate and high severity untreated areas. Forest floor fuels did not show a trend with increasing degree of treatment, but lessened with increasing burn severity. Crown base height, which is indicative of vertical fuel loadings and continuity, was highest in the cut and burn treatment and in higher-severity portions of all treatments. The abundance of the smallest size classes of trees in the diameter distribution is representative of ladder fuels; trees of these size classes were least abundant in the cut and burn treatment, high severity burn-only areas, and moderate and high severity untreated areas.

Management implications

The Rodeo-Chediski fire was undoubtedly extremely destructive, but was not as uniformly catastrophic as is often perceived, largely due to the fuel reduction treatments that were carried out before the fire. Yet, the recent treatments comprised less than a quarter of the burn area. It is a tragedy that while the White Mountain Apache Tribe has one of the oldest and most extensive prescribed burning programs in the West, huge portions of the burn area still experienced near-complete or complete mortality.

Our results suggest that recent prescribed burning can substantially increase crownfire resistance, however. This has been shown in several previous studies on White Mountain Apache Tribal lands over its long history of prescribed fire use (Weaver 1955, Kallander 1969, Finney et al., in press), as well as for wildfires occurring in other arid coniferous forests (Pollet and Omi 2002, Omi and Martinson 2002). When used alone, prescribed burning would have to be repeated frequently, which may present logistical and financial challenges; and, as mentioned earlier, it can be difficult to control and can kill many older trees (Sackett and Haase 1998). This study suggests that mechanical tree removal combined with prescribed burning offers the best resistance to catastrophic wildfire and subsequent type conversion away from ponderosa pine, even under the most extreme fire weather and proximate fire behavior. Although uneven-aged management differs from ecological restoration in many important characteristics, uneven-aged forest management including pre-commercial thinning or restoration to presettlement conditions (Covington et al. 2001), combined with prescribed burning, should both reduce overall tree density and the number of small trees enough to provide crownfire resistance. Similar types of fuel reduction have been recommended by many over the years (Weaver 1951; Agee and Skinner 2005).

Ponderosa pine will most likely lose dominance for several decades in most portions of the burn area while a thicket of oaks, New Mexico locust, and juniper matures and self-thins. Most treated areas should recover to a ponderosa pine/Gambel oak forest without further intervention, but due to very low levels of ponderosa regeneration, it may be necessary to do plantings in untreated areas to prevent a conversion to a shrubfield.

Unless widespread, regularly-applied fuel reduction projects are undertaken, most southwestern forests will remain highly susceptible to catastrophic wildfire. Climate change is expected to extend the fire season and possibly increase the number of large fires when droughts do occur, via a combination of increased temperature and precipitation (Swetnam and Betancourt 1990, Flannigan et al. 2001, McKenzie et al. 2004). Once these forests burn, they may take closer to centuries than decades to recover, and substantial portions of untreated areas may convert to oak-dominated shrubfields. We may see a great deal of fragmentation of what has been the largest contiguous ponderosa pine forest in the United States, with associated loss of timber revenue, and ecological effects that are not fully known. Outlying forests should not be completely neglected in favor of wildland/urban interfaces during decisions regarding fuel reduction priorities, for ponderosa pine forests face an uncertain future.

Table 3.1. Distribution of 1991-2001 treatments within the entire Rodeo-Chediski burn area and the sample area on White Mountain Apache Tribe lands. 111,837 hectares were burned in total on WMAT lands during the wildfire, and the total area sampled was 34,019 hectares.

Treatment	Entire burn area (ha)	Entire burn area (%)	Sample area (ha)	Sample area (%)
No treatment	78,072	69.81%	18,150	53.35%
Cut & prescribed burn	14,606	13.06%	10,747	31.59%
Prescribed burn only	10,466	9.36%	5,123	15.06%

Table 3.2. Overstory and regeneration species found on White Mountain Apache Tribe lands post-wildfire, all conditions. "% Presence" indicates the percent of plots on which the species was found. “Overstory” indicates a tree at least 1.4 m tall, and “Regeneration” indicates tree regeneration under 1.4 m.

Common name	**Scientific name**	**% Presence (Overstory)**	**% Presence (Regeneration)**
Ponderosa pine	*Pinus ponderosa*	94	24
Alligator juniper	*Juniperus deppeana*	29	17
Gambel oak	*Quercus gambelii*	27	60
Douglas-fir	*Pseudotsuga menziesii*	23	2
Gray oak	*Quercus grisea*	4	11
White fir	*Abies concolor*	4	< 1
Other oaks (unidentified)	*Quercus* spp.	3	10
Chihuahua pine	*Pinus leiophylla*	1	< 1
Utah juniper	*Juniperus osteosperma*	1	< 1
New Mexico locust	*Robinia neomexicana*	< 1	22
Southwestern white pine	*Pinus strobiformis*	< 1	0
Scrub (turbinella) oak	*Quercus turbinella*	< 1	5
Gambel/gray oak hybrid	*Quercus undulata*	< 1	2
Emory oak	*Quercus emoryii*	< 1	< 1
Black walnut	*Juglans nigra*	< 1	< 1
Bigtooth maple	*Acer grandidentatum*	< 1	0
Boxelder	*Acer negundo*	< 1	0
Two-leaf pinyon	*Pinus edulis*	< 1	0
Arizona white oak	*Quercus arizonica*	< 1	< 1
Willow	*Salix* spp.	< 1	< 1

Table 3.3. Shrub species found on White Mountain Apache Tribe lands post-wildfire, all conditions. "% Presence" indicates the percent of plots on which the species was found.

Commmon name	**Scientific name**	**% Presence**
Fendler's ceanothus	*Ceanothus fendleri*	67
Pinemat manzanita	*Arctostaphylos pungens*	21
Pringle manzanita	*Arctostaphylos pringlei*	7
Mountain mahogany	*Cercocarpus montanus*	6
Creeping barberry	*Mahonia repens*	5
Skunkbush sumac	*Rhus trilobata*	3
Other manzanita	*Arctostaphylos* spp.	2
Woods' rose	*Rosa woodsii*	2
Desert false indigo	*Amorpha fruticosa*	1
Chokecherry	*Prunus virginiana*	< 1
Obovate buckthorn	*Rhamnus betulifolia*	< 1

Table 3.4. DISTLM (Anderson 2004) was used to make univariate and multivariate comparisons across treatments, across severities, of interaction of treatment and severity, and pairwise by treatment. The Bray-Curtis dissimilarity distance measure was used, and the number of permutations for each test was 999. P-values and R^2 (proportion of variation explained) are below; non-significant *p*-values ($p > 0.05$) are italicized.

	Treatment		**Severity**		**Interaction**		**Burn only vs. Untreated**		**Cut & burn vs. Untreated**		**Burn only vs. Cut & burn**	
Characteristic	***P*-value**	**R^2**	***P*-value**	**R^2**	***P*-value**	**R^2**	***P*-value**	**R^2**	***P*-value**	**R^2**	***P*-value**	**R^2**
Density (trees/hectare)	0.002	0.01	0.001	0.32	0.050	0.01	0.022	0.01	0.002	0.02	0.029	< 0.01
Basal area	0.002	0.02	0.001	0.27	*0.063*	0.01	*0.055*	0.01	0.002	0.02	*0.152*	0.01
Crown base and bole char height, live	0.004	0.03	0.001	0.12	*0.243*	0.02	*0.86*	< 0.01	0.005	0.03	0.001	0.04
Bole char height, all	0.001	0.02	0.001	0.32	*0.103*	0.01	0.008	0.01	0.001	0.03	*0.251*	< 0.01
Diameter distribution	0.002	0.01	0.001	0.20	*0.131*	0.01	0.027	0.01	0.002	0.02	0.029	0.01
Snags	0.001	0.05	0.001	0.10	*0.979*	< 0.01	0.024	0.01	0.001	0.07	0.003	0.03
Fuels, grouped by forest floor/fine/coarse	0.034	0.01	0.001	0.06	0.005	0.02	*0.5*	< 0.01	0.004	0.01	*0.183*	0.01
Regeneration (unstandardized)	0.001	0.02	0.001	0.02	0.024	0.01	*0.124*	0.01	0.001	0.02	0.001	0.02
Regeneration (standardized by species)	0.001	0.02	0.001	0.01	0.005	0.01	0.04	0.01	0.001	0.03	0.001	0.02
Pinus ponderosa	0.001	0.06	*0.244*	0.01	0.014	0.02	0.001	0.04	0.001	0.09	0.009	0.02
Quercus gambelli	*0.099*	0.01	0.001	0.03	*0.650*	0.01	*0.185*	< 0.01	*0.068*	0.01	*0.172*	0.01
Robinia neomexicana	*0.399*	< 0.01	0.005	0.02	0.015	0.02	*0.512*	< 0.01	*0.166*	0.01	*0.602*	< 0.01
Shrubs (unstandardized)	0.001	0.03	0.001	0.03	0.004	0.02	0.001	0.02	0.002	0.01	0.001	0.04
Shrubs (standardized by species)	0.001	0.02	0.001	0.03	0.006	0.02	0.001	0.02	0.001	0.01	0.001	0.03
Ceanothus fendleri	0.001	0.03	0.001	0.04	0.003	0.02	0.001	0.03	*0.216*	< 0.01	0.001	0.04
Arctostaphylos spp.	0.019	0.01	0.002	0.03	*0.217*	0.01	*0.395*	< 0.01	0.001	0.03	0.006	0.02

Figure 3.1. Burn severity as measured by ΔNBR over the extent of the Rodeo-Chediski fire was mixed. The burn area and White Mountain Apache Tribe lands in relation to the state of Arizona are shown in the lower left. The "low" severity category included unburned areas.

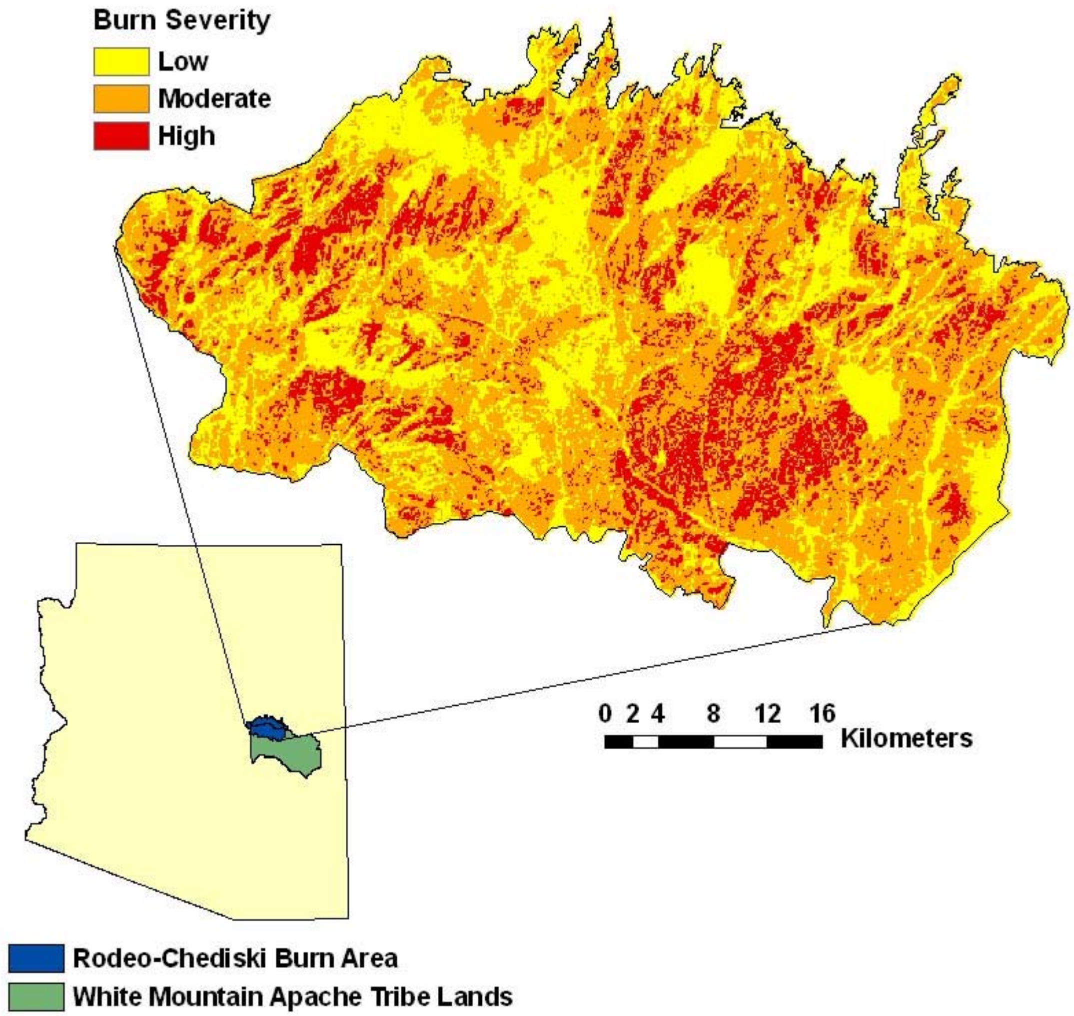


Figure 3.2. Comparing the burn severity distribution after the Rodeo-Chediski fire for areas treated with prescribed burning over increasing treatment periods indicates that 1991-2001 is the longest period for which the burn severity distribution is substantially different from that of the entire burn area; this is confirmed by a Kolmogorov-Smirnov test of the raw ΔNBR distributions for the treatment periods with α= 0.01. Missing years in the sequence are those in which no prescribed burning took place.

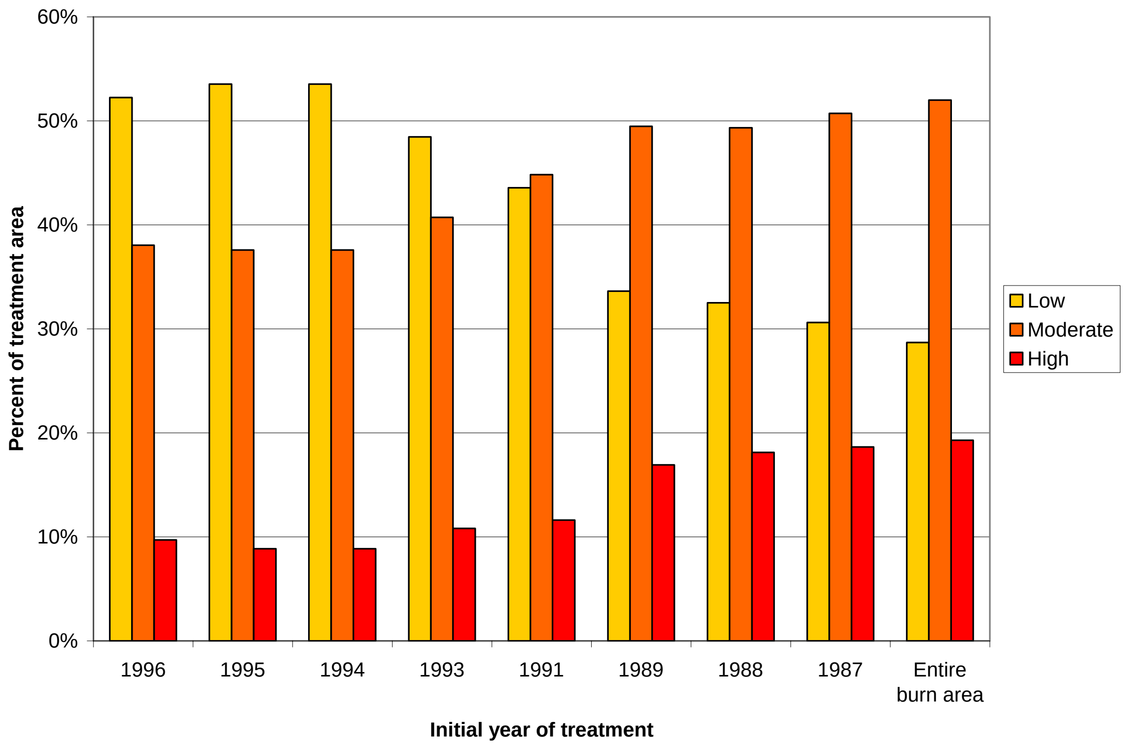


Figure 3.3. The burn severity distribution after the Rodeo-Chediski fire for areas that underwent fuel reduction treatments between 1991 and 2001 indicates that the combination of cutting and prescribed burning had the greatest ameliorative effect on burn severity, though prescribed burning alone also considerably reduced burn severity as compared to untreated areas and the burn area on WMAT lands as a whole. All pairwise comparisons of the severity distributions for each treatment type are significantly different (Kolmogorov-Smirnov test of the raw ΔNBR distributions for the treatment types with $\alpha = 0.01$).

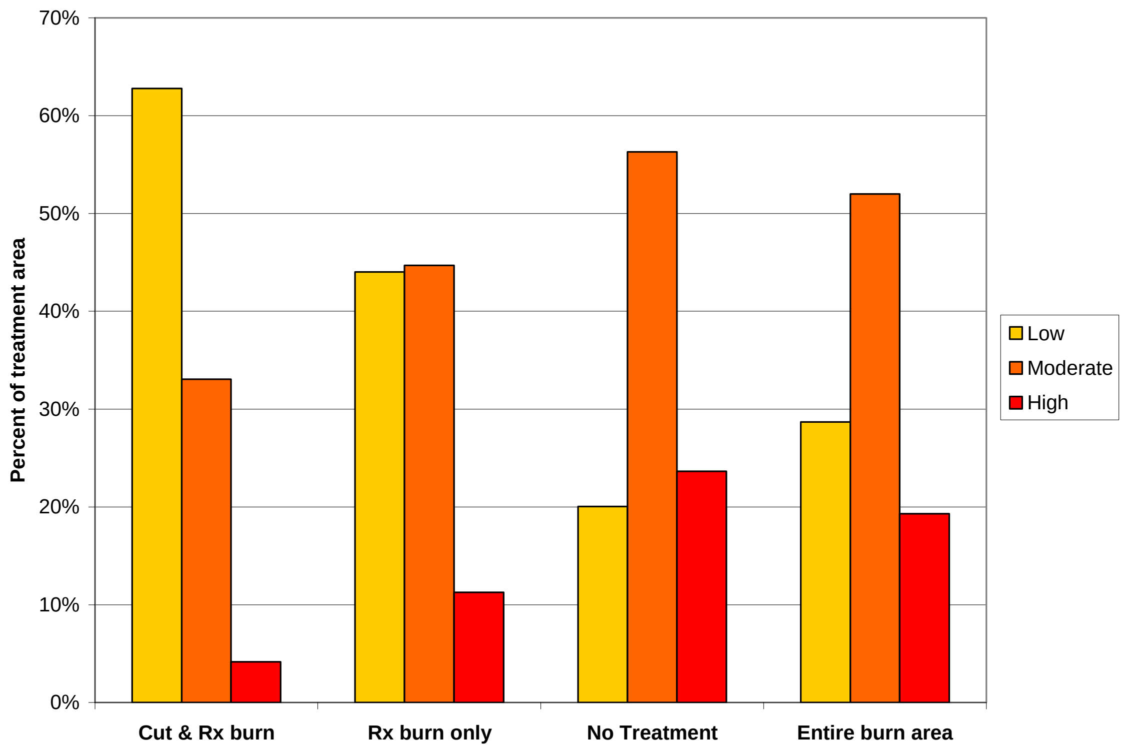


Figure 3.4. Study sites on the Rodeo-Chediski burn area within White Mountain Apache Tribal lands, with the area sampled: 1991-2001 treatments and areas ≥ 2000 m in elevation and ≤ 45% slope. Area sampled represents 30% of the total burn area within WMAT lands.

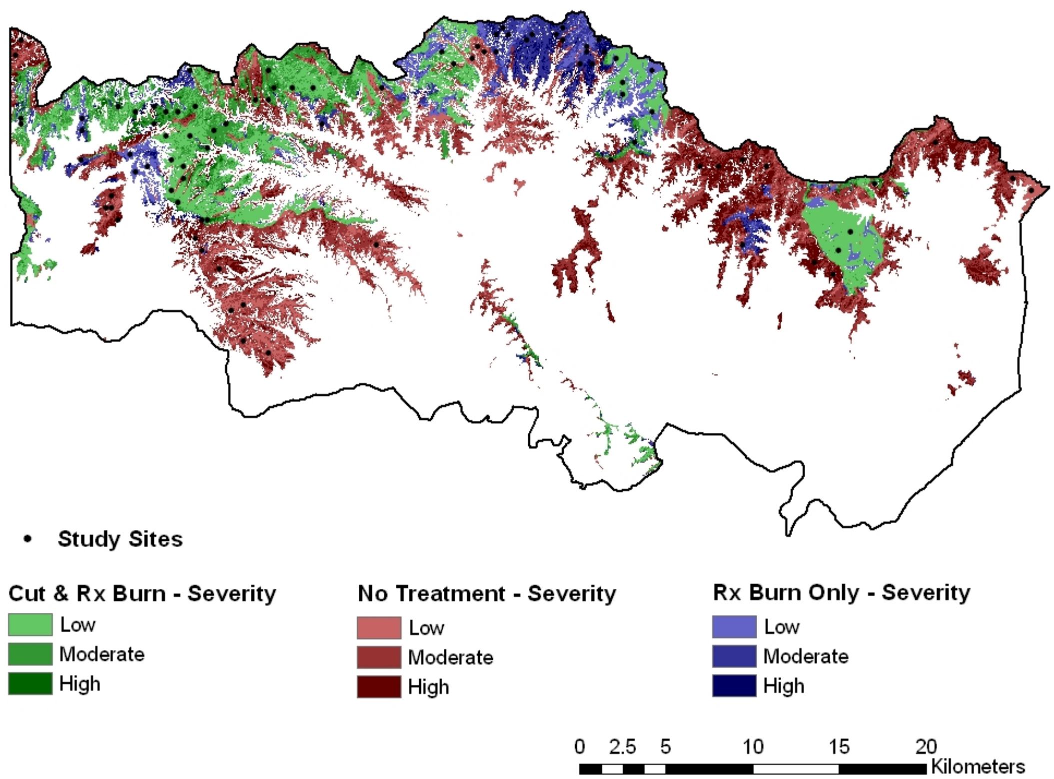


Figure 3.5. Forest density decreases with increasing burn severity ($p = 0.001$), and mortality appears to be highest in areas with no pre-fire fuel reduction treatment ($p = 0.002$ for overall treatment effect), but interaction of treatment and severity was significant ($p = 0.05$). All pairwise treatment comparisons were significantly different. Error bars for this and all following figures represent ±1 standard error.

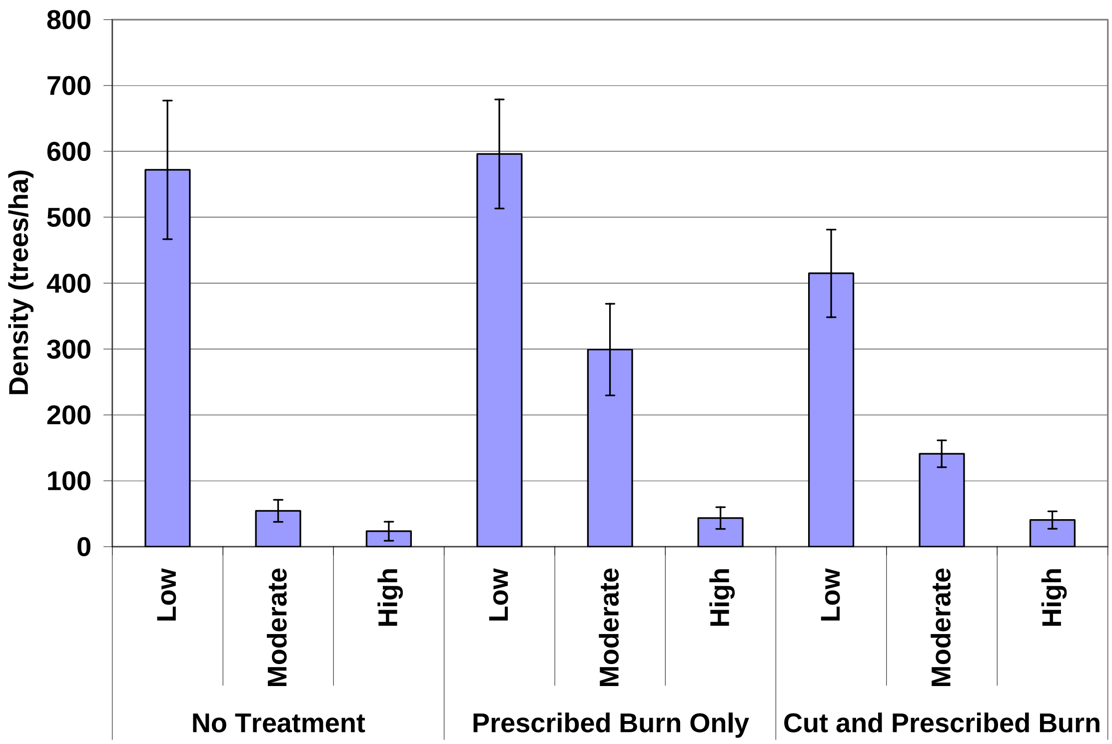


Figure 3.6. Basal area decreases with increasing burn severity (p = 0.001), and increases with the degree of treatment for moderate and high burn severities (p = 0.001 for overall treatment effect). Basal area on cut and burned vs. untreated areas was different (0.002), but was not significantly different for other pairwise treatment comparisons.

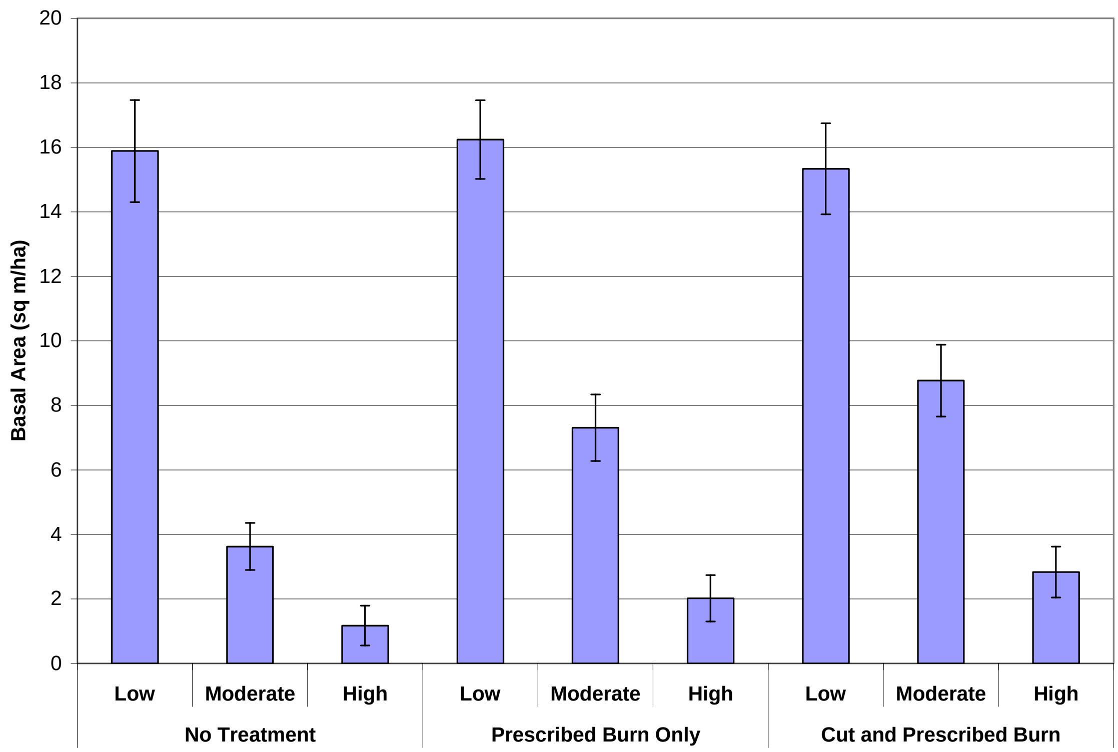


Figure 3.7. Crown base height and bole char height generally increase with increasing burn severity ($p = 0.001$). Crown base height is highest in the cut and prescribed burn treatment ($p = 0.004$ for overall treatment effect), while bole char height is highest in areas with no treatment ($p = 0.001$) However, there was a significant interaction effect for both crown base and bole char ($p = 0.243$ and 0.103, respectively).

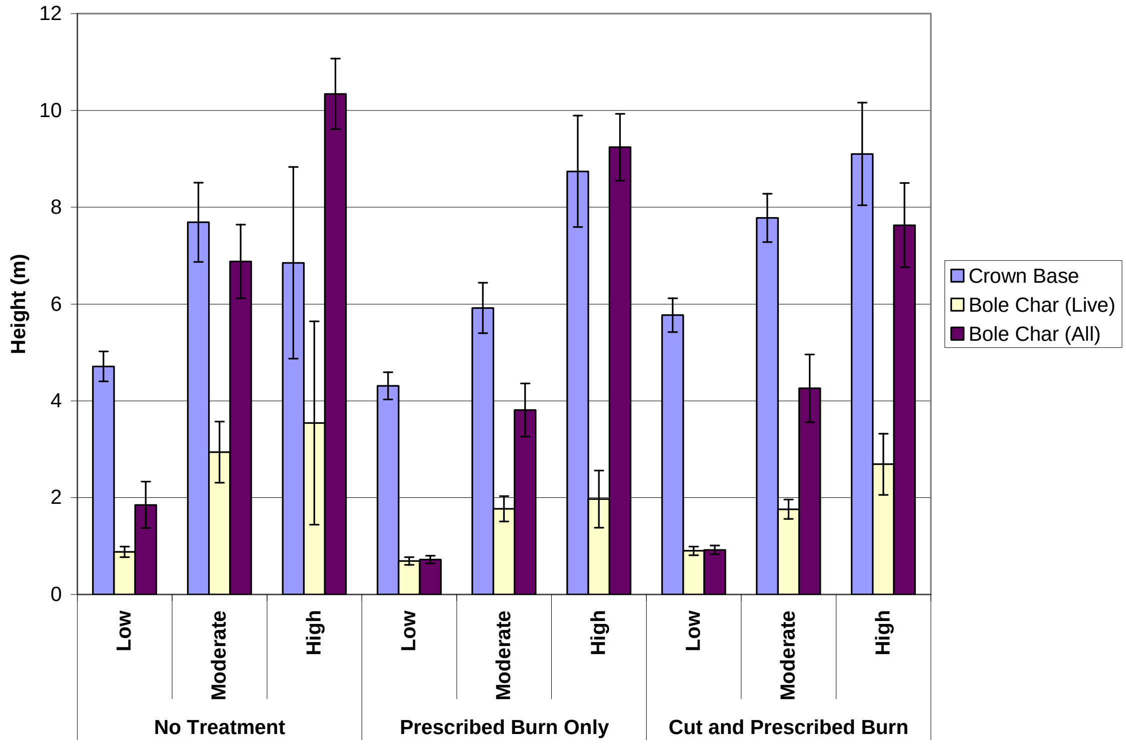


Figure 3.8. The diameter distribution for ponderosa pine was significantly affected by treatments ($p = 0.002$) and by burn severity ($p = 0.001$). Far fewer small trees are present as burn severity increases, especially in untreated areas. All pairwise treatment comparisons are significantly different.

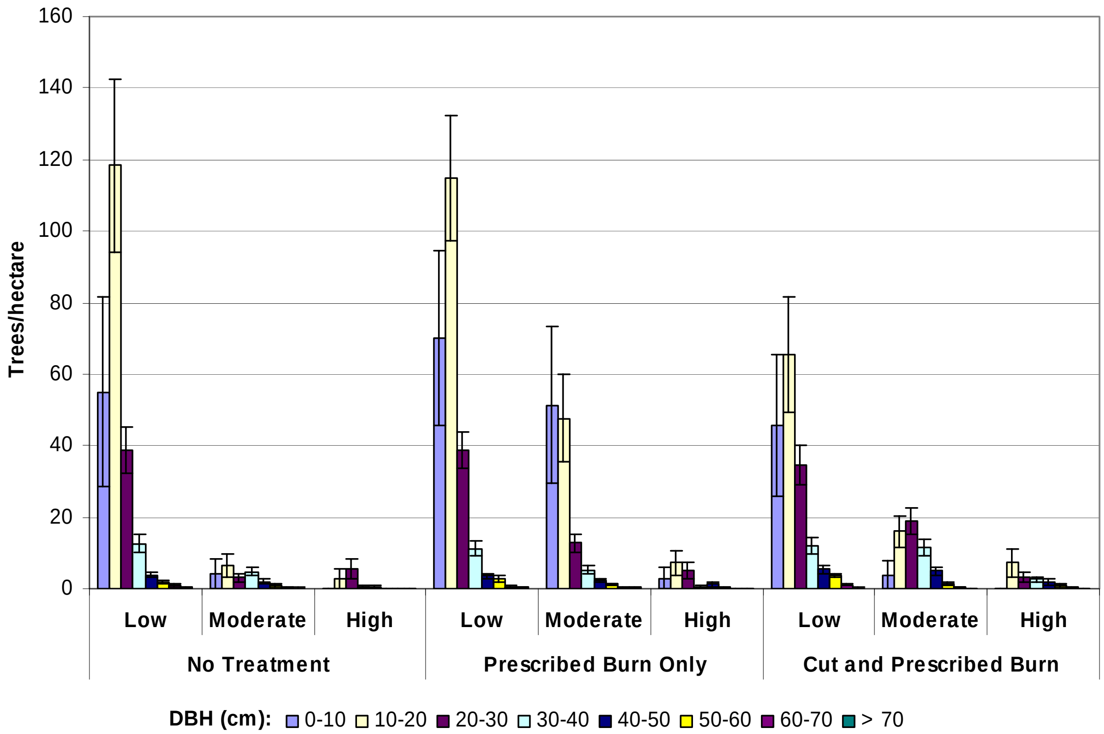


Figure 3.9. Cumulative age frequency for all species. The vast majority of trees, regardless of treatment or severity, are less than 100 years old. Treated areas appear to have more old trees, especially as burn severity increases. NT = no treatment; Rx = burn only; CB = cut and burn.

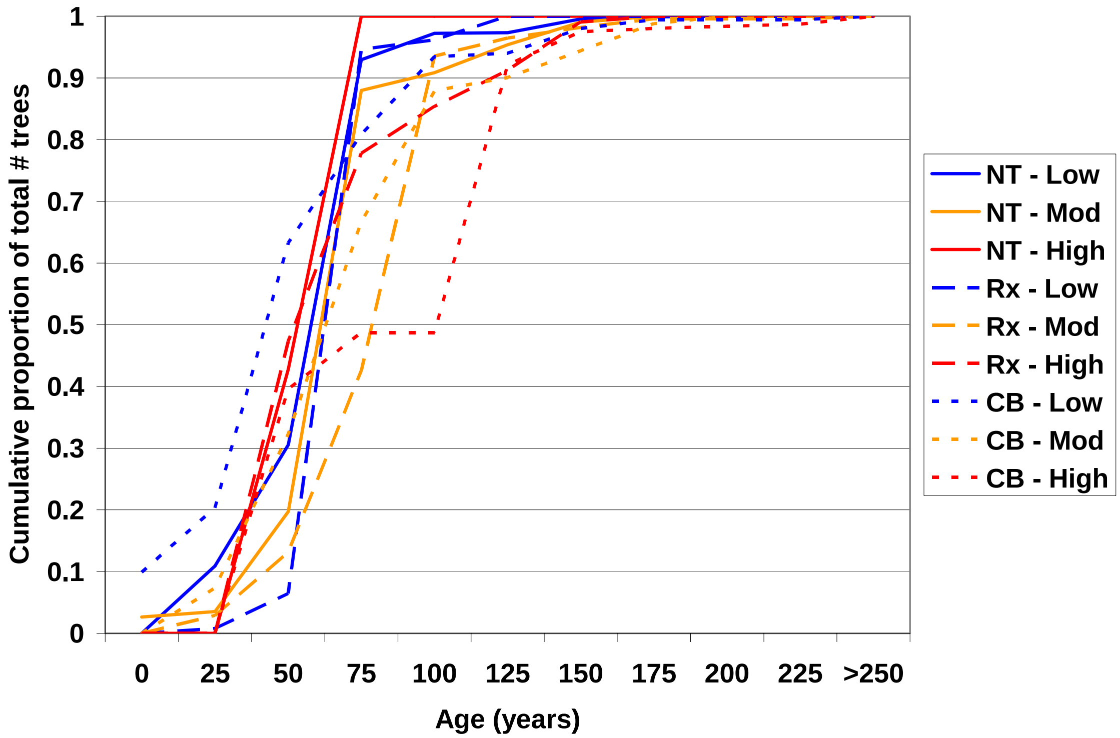


Figure 3.10. Snag density of two common size classes used by wildlife (>30 cm and >50 cm DBH) increased with burn severity ($p = 0.001$) and decreased with degree of treatment ($p = 0.001$).

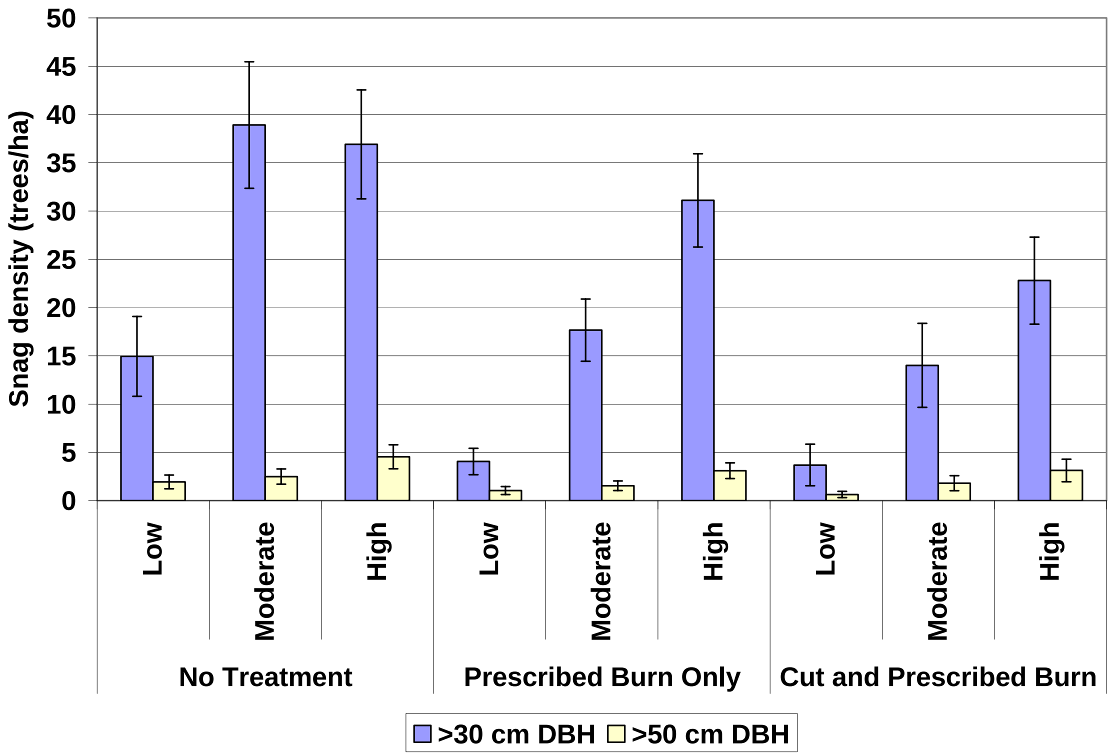


Figure 3.11. Fuel loadings decreased somewhat with increasing burn severity (0.001); this trend appeared to be dominated by changes in forest floor fuels. However, there was a significant interaction effect of treatment and severity ($p = 0.005$).

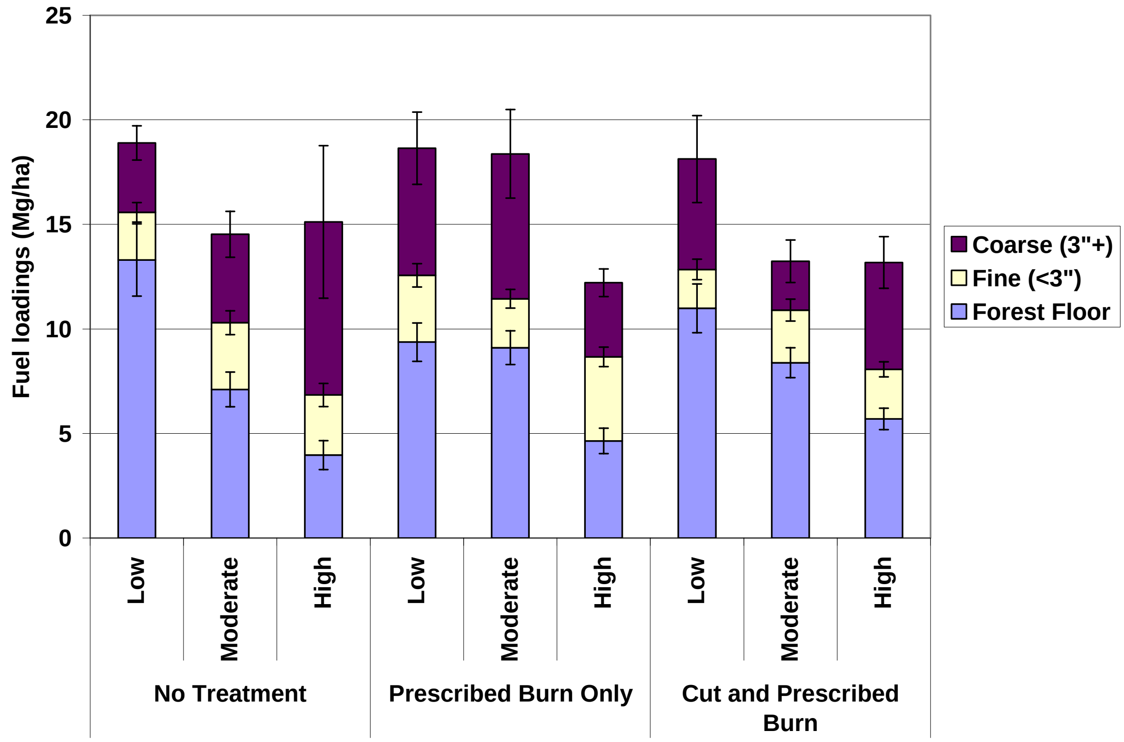
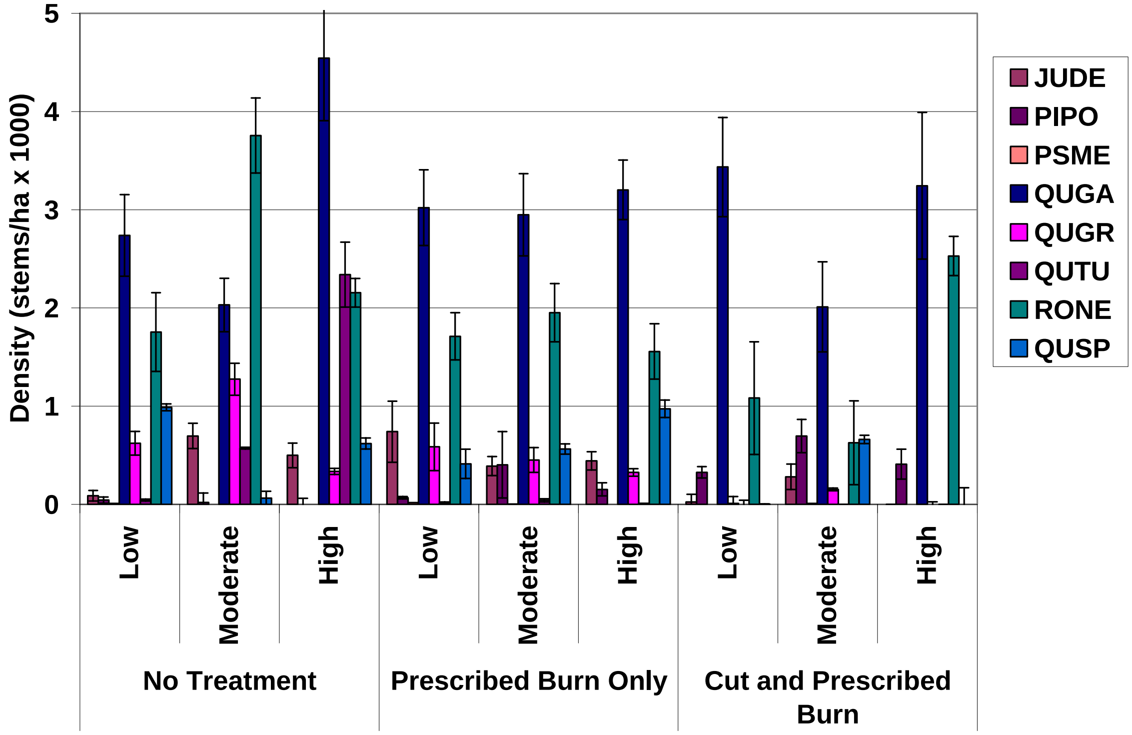


Figure 3.12. Post-wildfire regeneration was dominated by Gambel oak and New Mexico locust, and was subject to a significant interaction effect ($p = 0.024$). Only species found on at least 2% of plots are shown, and less-common *Quercus* species are grouped as "QUSP".



JUDE = *Juniperus deppeana*, PIPO = *Pinus ponderosa*, PSME = *Pseudotsuga menziesii*, QUGA = *Quercus gambelii*, QUGR = *Quercus grisea*, QUTU = *Quercus turbinella*, RONE = *Robinia neomexicana*, QUSP = Other *Quercus* species, grouped.

Figure 3.13. Ponderosa pine regeneration responded most strongly to moderate burn severity in treated areas, and there was no regeneration in untreated areas of high severity. Overall treatment effect (p = 0.001) and all pairwise treatment comparisons were significant.

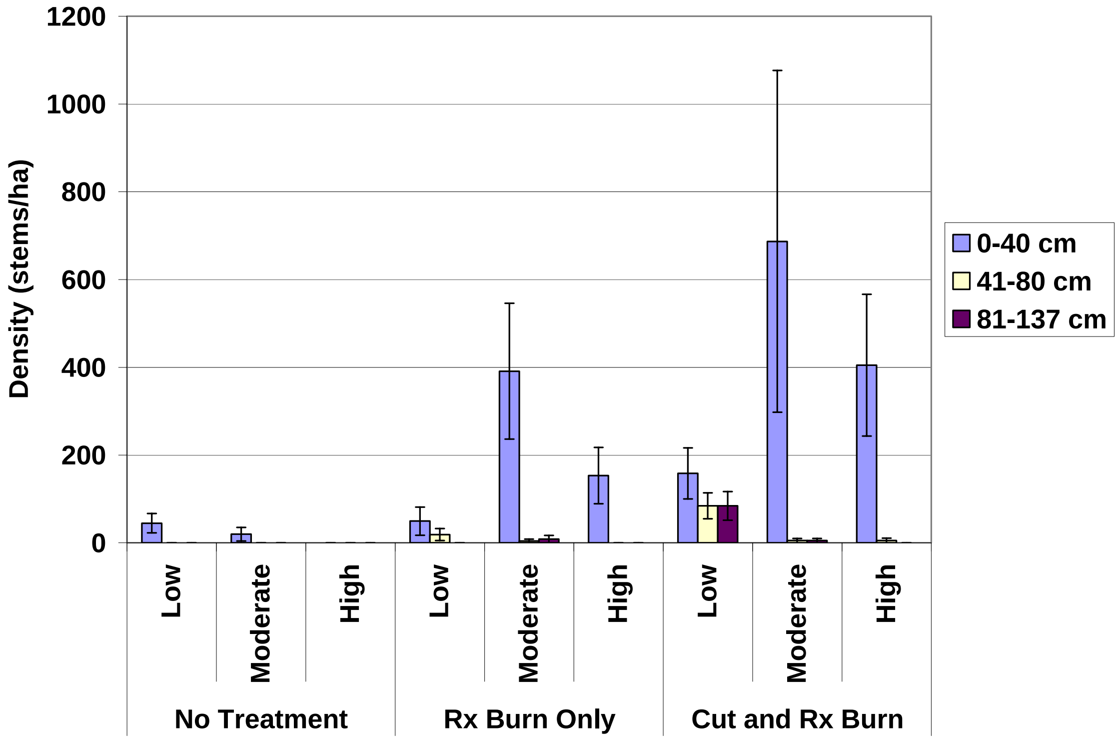
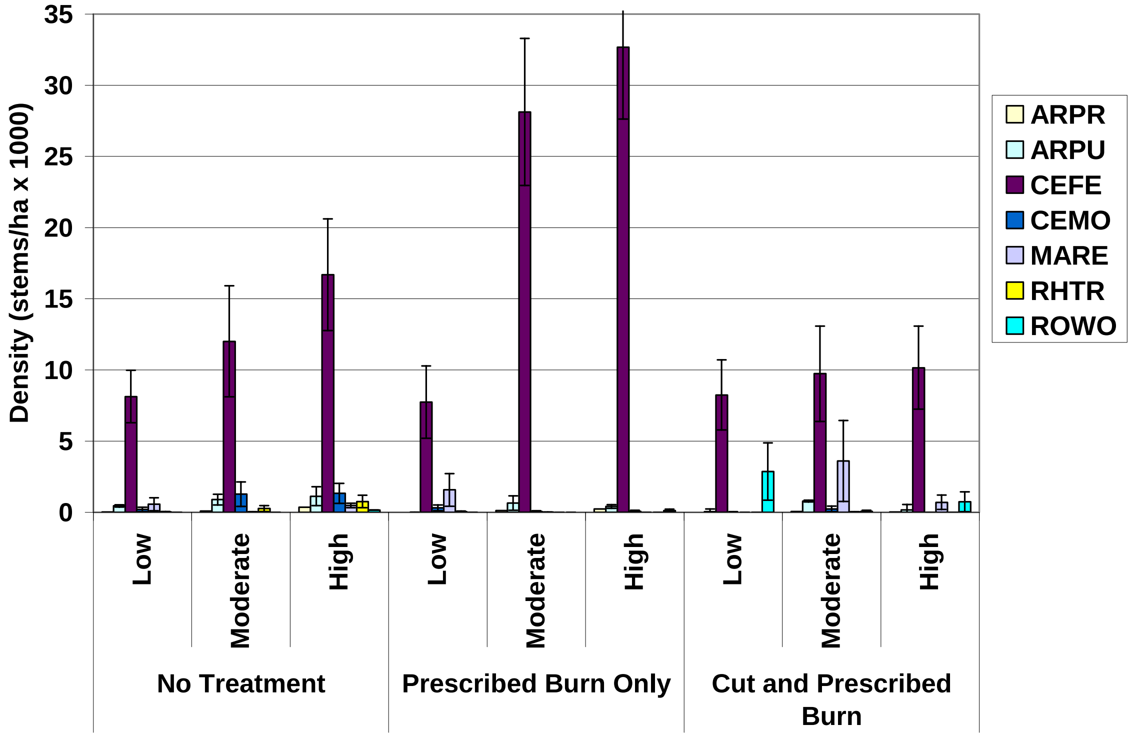


Figure 3.14. *Ceanothus fendleri* was by far the most common shrub, increasing in density with increasing burn severity and in the "burn only" treatment. While treatment and severity effects were significant for overall shrubs and *C. fendleri* considered separately ($p = 0.001$ in all cases), there was also an interaction effect ($p = 0.004$ and 0.003 for overall shrubs and *C. fendleri,* respectively). Only species occurring on at least 2% of plots are shown.



ARPR = *Arctostaphylos pringlei*, ARPU = *Arctostaphylos pungens*, CEFE = *Ceanothus fendleri*, CEMO = *Cercocarpus montanus*, MARE = *Mahonia repens*, RHTR = *Rhus trilobata*, ROWO = *Rosa woodsii*.

4. PRE-FIRE TREATMENT EFFECTS AND POST-FIRE FOREST DYNAMICS ON THE APACHE-SITGREAVES NATIONAL FOREST WITHIN THE RODEO-CHEDISKI BURN AREA, ARIZONA

Abstract

The 2002 Rodeo-Chediski fire was the largest wildfire in Arizona history, and exhibited some of the most extreme fire behavior ever seen in the Southwest. On the portion of the Apache-Sitgreaves National Forest within the burn area, pre-fire thinning set the stage for a natural experiment testing the upper boundary of effectiveness of fuel reduction treatments at decreasing burn severity. We sampled seven pairs of thinned/unthinned stands two years after the fire. Thinned areas had more live trees, higher survival, and less extreme fire behavior as indicated by crown base height and bole char height. Ponderosa pine regeneration was patchily distributed and somewhat less abundant in untreated areas. Our findings strongly indicate that thinning was associated with reduced burn severity even in an extraordinarily intense fire. Differences between thinned and untreated areas persisted for several decades after the fire in stand structure characteristics and for at least 100 years in species composition when modeled using the Forest Vegetation Simulator. Future forest development will most likely take one of two trajectories: recovery to a ponderosa pine/Gambel oak forest or a shift to an alternative stable state such as an oak-dominated shrubfield, with untreated areas more apt to undergo a shift to a shrubfield state.

Introduction

There has been extensive empirical research on the long-term recovery of pine-dominated forests after wildfire (Foxx 1996, Barton 2002, Gracia et al. 2002, Greene et al. 2004, Savage and Mast 2005), and a number of studies have also modeled forest dynamics after wildfire (He et al. 2002, Retana et al. 2002, Chapin et al. 2003). Several of these long-term studies of forest recovery after wildfire have questioned whether forests that historically had a frequent fire regime are resilient to crownfire, adding to the widespread concern about future development of arid forests such as those of the Southwest as large, severe crownfires continue (Hessburg et al. 2005). Studies in Arizona (Barton 2002), Mexico (Fulé et al. 2000), and Spain (Retana et al. 2002) have indicated that intense fire in pine-oak forests may result in a shift to a more oak-dominated forest or a conversion to an alternative steady state such as a shrubfield. These shifts in species dominance and conversions to what appears to be an alternative stable state have been documented after crownfire in several dense southwestern ponderosa pine forests by Savage and Mast (2005). Alternative stable states are self-perpetuating species assemblages distinct from the typical assemblage found in a given environment; they may be shorter in height (e.g., shrubs vs. trees) and appear to be an earlier sere, though without evidence of near-term shift back toward the pre-crownfire forest. A shift to such an alternative state could have major consequences for ecosystem functions and potential land uses; for instance, in the Southwest, some wildlife species are ponderosa pine-dependent, and Gambel oak is much less economically valuable than the ponderosa pine it could replace as the dominant tree species (Barger and Ffolliott 1972, Blatner and Govett 1988, Linhart 1988, Petraitis and Latham 1999).

Changes in forest structure caused by fire suppression, selective logging of large trees, and livestock grazing occurring after extensive EuroAmerican settlement of the western U.S. have led to an increase in the size and frequency of crownfires (Covington and Moore 1994, Swetnam et al. 1999). Fuel reduction treatments such as pre-commercial thinning, prescribed burning, and restoration to presettlement conditions have been projected to decrease crownfire susceptibility by decreasing forest floor, ladder , and canopy fuels (Deeming 1990, van Wagtendonk 1996, Stephens 1998, Covington et al. 2001, Fulé et al. 2001, Brose and Wade 2002). Numerous short-term studies have examined adjacent treated and untreated stands after wildfire, reporting that areas that underwent fuel reduction treatments experienced lower burn severity (Wagle and Eakle 1979, Omi and Kalabokidis 1991, Vihanek and Ottmar 1993, Agee et al. 2000, Martinson and Omi 2003). Pollet and Omi (2002) and Omi and Martinson (2002) systematically examined the effect of fuel reduction treatments for eight severe wildfires, both studies finding that burn severity and crown scorch were lower in treated areas. Cram and Baker (2003) also systematically investigated silvicultural treatment effect after four wildfires, including the Rodeo-Chediski burn area within the Apache-Sitgreaves National Forest, finding that treated areas up to 20 years old experienced lower burn severity and estimated fireline intensity. Pollet and Omi (2002) questioned whether fuel treatments will have any effect under extreme fire conditions, since drought and high winds may play a more important role in fire behavior than fuels. By decreasing burn severity and thus preventing long-term forest change, fuel reduction treatments can potentially have enduring impacts (Savage and Mast 2005).

Because of its extreme fire behavior, which included multiple plume collapses per day, flame lengths of 60-120 m (200-400 ft), and a maximum rate of spread of 6.4 km/hour (4 mph; USFS 2002), the Rodeo-Chediski fire can serve as a test case of the upper boundary of response to fuel treatments. In addition, little research has been done investigating the effect of fuel reduction treatments on burn severity and projected future growth, including potential shifts to an alternative stable state, across treated and untreated areas. Our hypotheses were that in an extraordinarily intense fire (a) pre-wildfire thinning led to lower burn severity than in untreated areas; (b) post-fire regeneration and shrubs, and thus potential forest development, differs on thinned and untreated areas; (c) differences in post-fire recovery of thinned and untreated areas will persist for several decades following the fire as projected by a vegetation simulation model; and (d) untreated areas are more likely to transition from the pre-fire ponderosa pine forest to an oak-dominated shrubfield.

Methods

Study Area

The Rodeo-Chediski burn area's 189,000 hectares (468,000 acres) span the Mogollon Rim in east-central Arizona. This study focused on the portion of the burn area on the Apache-Sitgreaves National Forest, to the north of the Mogollon Rim. July maximum temperature is 29.2ºC (84.5ºF), January minimum temperature is -7.8ºC (17.9ºF), annual precipitation is 50.6 cm (19.9 in), and annual snowfall is 99.3 cm (39.1 in); these are 1971-2000 averages (excepting snowfall, a 1950-2004 average) from the Heber Ranger Station on the northwestern edge of the burn area (Western Regional Climate Center, www.wrcc.dri.edu). The study sites ranged in elevation from 1990 – 2138 m (6,530 – 7,015 ft). The soil type varies from clay substrates to sandy loams, depending on the parent material; alluvial gravels are present in drainages, and the Mogollon Rim itself has a limestone bed. Forests were dominated by ponderosa pine (*Pinus ponderosa*) with Gambel oak (*Quercus gambelii*), alligator juniper (*Juniperus deppeana*) and New Mexico locust (*Robinia neomexicana*).

Study Sites

Seven pairs of thinned and unthinned stands were sampled on the Apache-Sitgreaves National Forest in May-August of 2004 (Figure 1). Six of these pairs and one unpaired thinned site were pre-existing sites sampled immediately post-fire by the Forest Service (USFS 2002, L. Wadleigh and C. Hoffman, pers. comm 2003). A seventh unthinned site was located near the pre-existing unpaired thinned site. Thinned sites underwent pre-commercial thinning followed by slash disposal (varying from lop and scatter to pile and burn) as part of commercial timber operations between 1990 and 1999. Two systematic grids of five plots each were established on each study site, for a total of ten plots per site; 140 plots in total were measured across the entire study area. These plots did not correspond exactly to those sampled by the Forest Service, but the areas sampled were roughly equivalent. Plots were located using a Garmin GPS 12 with ten-meter resolution. If the original location of a plot fell on a slope of greater than 45%, the plot center was moved to the nearest area where the majority of the plot would be less than 45% slope. The slope constraint ensured that we would not be comparing treatable slopes to areas that could not have feasibly been treated because they were on unworkably steep slopes.

Measurements

Overstory trees were measured on a variable-radius plot using a prism with a BAF of approximately 2.2 m^2/hectare per tree (10 ft^2/acre per tree). Tree measurements included: tree species, condition, diameter at breast height (~1.4m), total height, canopy base height, bole char height (minimum and maximum), and dwarf mistletoe rating (Hawksworth 1977). Tree condition classes followed Thomas' (1979) description and included: live, declining, and four stages of snags (recent snag, loose-bark snag, clean snag, and snag broken above breast height). Since field measurements were taken two years after the fire, we did not attempt to estimate foliage scorch, as the majority of scorched needles had already fallen from the trees. A subsample of trees (the live specimens of the first four trees on the plot; trees were numbered starting at north and proceeding clockwise around the plot) were cored in order to produce age and growth increment data. Tree increment cores were surfaced and crossdated (Stokes and Smiley, 1968) following standard methods or rings were counted for cores that could not be crossdated, e.g., some junipers. For cores that missed the pith, additional years to the center were estimated with a pith locator (Applequist 1958). Ten-year diameter growth increments were measured (1994-2003, inclusive).

Tree regeneration (saplings and seedlings below breast height) and shrubs were measured on a 40.5 m^2, 3.6 m radius plot (1/100 acre) with origin at plot center. Tree regeneration and shrubs were tallied by species, condition (living or dead), and height class (0-40 cm, 41-80 cm, 81-137 cm, or exact height if > 137 cm). We measured forest floor fuels on a 15.24-meter (50-foot) planar transect at a random azimuth from each plot center, using Brown's (1974) method. Coefficients for planar transect calculations are from Sackett (1980). Coefficients used to convert litter and duff depth to forest floor fuel loadings in megagrams/hectare are from Ffolliott et al.'s (1968) measurements of forest floor weight in northern Arizona ponderosa pine stands.

We took two photographs of each plot: a hemispherical photo using a digital camera with a 180º fisheye lens (Nikon CoolPix E4300 using FC-E8 Fisheye Converter Lens with UR-E4 Converter Adapter) at plot center in order to record canopy cover, and a plot photograph using a standard lens (Canon PowerShot A70) from 12 m east of plot center. Hemispherical photos were analyzed with Gap Light Analyzer (Institute of Ecosystem Studies 1999) in order to determine forest canopy structure and gap light transmission indices. An iron rebar stake was sunk to ground level at each plot center and tagged with the plot number as a permanent marker.

Modeling

We used the Central Rockies/Southwestern Ponderosa Pine variant of the Forest Vegetation Simulator (FVS), an individual-tree growth and yield statistical model (Dixon 2003) to project future growth on thinned and untreated sites. FVS is initialized with standard mensurational data, and outputs both stand-level and tree-level growth data which can be analyzed by species. FVS can be quite accurate over relatively short simulations especially due to its tailoring to specific ecosystems; this variant was based on the GENGYM model (Edminster et al. 1991) and is customized such that its projected results are in accordance with known stand dynamics of the area.

FVS was used to simulate stand development for each thinned and untreated site for the next 100 years (2004-2104). Ten-year growth increments were used to scale the diameter increment model, and the site index was reduced from the default of 21.4 m/100 years (70 ft/100 years) to 15.2 m/100 years (50 ft/100 years) to correspond with height/age relationships developed using our core data. All oaks were treated as Gambel oak, all junipers were treated as a generic juniper species, all pines were treated as ponderosa pine and New Mexico locust was treated as a generic hardwood species for the purposes of the simulation.

We compared two regeneration scenarios:

Regen-1: Establishing the measured regeneration in 2004, with no further regeneration established over the course of the simulation; and

Regen-2: Establishing the measured regeneration in 2004, and in 2024, establishing the measured regeneration scaled by regeneration occurring 2-3 decades following crownfire in southwestern ponderosa pine forests as reported by Savage and Mast (2005). Five of the ten burn areas they studied either overlapped or were within approx. 100 km of the Rodeo-Chediski burn area; these included three of the four burn areas they measured 2-3 decades post-fire. Juniper regeneration was decreased by a factor of 0.15 so that our average juniper abundance matched that of Savage and Mast, and oak regeneration was similarly decreased by a factor of 0.65. Since the range in abundance of New Mexico locust on our study sites was similar to that reported by Savage and Mast, we did not alter regeneration density for that species. It would have been necessary to increase our measured ponderosa regeneration by a factor of 28 to reach Savage and Mast's average, and we did not believe such a large increase was justifiable. Instead, ponderosa pine density was tripled, and on sites where there were less than 50 stems/ha of the smallest height class, 150 stems/ha were introduced. This did not increase ponderosa pine regeneration density established in 2024 to a level equivalent with the average reported by Savage and Mast (approx. 950 stems/ha), but it fell within the range they reported (an overall average of 212 stems/ha, compared with their range of 117-2864 stems/hectare).

Expected survival percentages for each species and height class for both regeneration scenarios were estimated using regeneration measured in untreated areas of two northern Arizona ponderosa pine forests (Fulé et al. 2001, Waltz et al. 2003). For each species, the abundance of individuals of the tallest height class relative to the abundance of individuals of a shorter height class was used as an estimate of the survival rate for individuals of the shorter height class. Ponderosa pine survival was 100% for all height classes, oak survival ranged from 9-45%, juniper survival ranged from 19-67%, and New Mexico locust survival ranged from 65-72%.

Data Analysis

While our post-fire data were parametric, inspection of the measurement distributions using frequency histograms, the Shapiro-Wilkes W statistic, and Levene's test revealed that only a few measurements were both normally distributed and had equal variances. We used DISTLM (Anderson 2004), which performs a distribution-free, distance-based multifactor multivariate analysis of variance using permutation. We carried out 999 permutations for each test, and used the Bray-Curtis dissimilarity distance measure for each test. The alpha used to denote a significant difference was 0.05. Since the method used by DISTLM calculates an exact *p*-value, it has been argued that it is not subject to alpha inflation (Anderson 2001, Anderson and Robinson 2001). When testing overall regeneration and shrubs, we performed both an unstandardized test and a test standardized by species, in case the dominant species were skewing the results.

We used 95% confidence intervals to discern meaningful differences between modeling results for thinned and untreated areas, and between the two regeneration scenarios.

Results

The burn area was dominated by ponderosa pine, though Gambel oak and alligator juniper were also common in the overstory (Table 4.1). Pinemat manzanita (*Arctostaphylos pungens*) and Fendler's ceanothus (*Ceanothus fendleri*) were by far the most abundant shrubs (Table 4.2).

Pre- and post-fire forest structure

Thinned areas were associated with lower burn severity when compared to untreated areas. Pre-fire forest structure in thinned and untreated areas was not significantly different according to the characteristics we tested, but post-fire conditions were very different (Table 4.3). There were many more surviving trees in thinned areas, and fire behavior was less extreme than in untreated areas as indicated by bole char height.

About half the trees in thinned areas survived the fire, compared to five percent in untreated areas (Figure 4.2a). Post-fire density of live trees was significantly higher in thinned areas. While average tree density was greater in untreated areas before the fire, there was no significant difference between thinned and untreated areas. At least some trees survived on all of the study sites, but 39% of the plots (10 plots per study site) had no live trees; by treatment, this was 66% of untreated plots and 11% of thinned plots. Basal area of live trees was similar before the fire, but was significantly higher in thinned areas afterwards (Figure 4.2b). Quadratic mean diameter before the fire was 14 cm in untreated areas and 18.5 cm in thinned areas. After the fire, quadratic mean diameter increased to 25 cm in untreated areas and 22 cm in thinned areas.

Crown base height and bole char height for live trees and for all trees was significantly lower in thinned areas, indicating less extreme fire behavior (Figure 4.2c). Bole char height for all trees was much higher than bole char for live trees and somewhat higher than crown base height in untreated areas, probably indicating that most trees that survived were in pockets of lower than average fire intensity. The fire probably did not increase crown base height in thinned areas, since crown base height was substantially higher than bole char height for all trees. It appears to have done so in untreated areas, especially since crown base height there was higher than in thinned areas.

Diameter distribution before the fire was not significantly different between treatments, though on average there were more small trees in untreated areas (Figure 4.2d). The fire shifted the distribution in untreated areas strongly towards larger trees, but did not substantially change the shape of the distribution in thinned. About 95% of the surviving trees in both thinned and untreated areas were less than 100 years old, though thinned areas may have had slightly more old trees (Figure 4.2e). Many more standing snags were present in untreated areas of two common size classes used by wildlife (>30 cm DBH and >50 cm DBH; Figure 4.2f).

Initial post-fire recovery

Initial post-fire recovery as denoted by fuel loadings, regeneration, and shrubs was more similar across thinned and untreated areas than post-fire forest structure. Only fuel loadings and manzanita density were significantly different. Fuel loadings were significantly higher in thinned areas (Figure 4.3a). This difference was apparent in fine and coarse woody debris as well as forest floor weight

Regeneration was dominated by sprouting species such as Gambel oak and alligator juniper, as well as by New Mexico locust (Figure 4.3b). Regeneration levels were not significantly different between thinned and untreated areas, but on average were slightly higher in untreated areas. Regeneration was present on every study site and 86% of the plots. Oak regeneration was present on all study sites and 68% of the plots.

There was no significant difference in ponderosa pine regeneration between treatments because of its patchiness, but on average, there was four times as much ponderosa pine regeneration in thinned areas (Figure 4.3c). Ponderosa pine regeneration was found on only six percent of our plots; at the site level, eight of 14 sites (5 untreated, 3 thinned) had no ponderosa regeneration. In both thinned and untreated areas, small amounts of ponderosa regeneration in the larger height classes may have survived the fire. Live ponderosa trees were present on all sites, but 41% of the plots (67% untreated, 14% thinned) had no surviving ponderosa pine trees. Thirty-eight percent of the plots had no surviving ponderosa pines and no ponderosa pine regeneration; by treatment, this was 64% of untreated plots and 11% of thinned plots.

Pinemat manzanita and Fendler's ceanothus were virtually the only shrubs present (Figure 4.3d). While overall, shrubs were not significantly different on thinned versus untreated areas, manzanita was twenty times more abundant in untreated areas than thinned areas ($p = 0.043$),. We thought it worthwhile to investigate whether manzanita may have been competing with ponderosa pine regeneration. Manzanita was recorded on

18 and ponderosa pine regeneration on nine of 140 plots; they were only found together on one plot. However, this is not a conclusive finding because the number of plots on which either species was found was so low..

Modeling

Differences between thinned and unthinned areas will likely persist for at least the next several decades in terms of overall forest structure characteristics as indicated by our modeling of future forest development using FVS, though variability was high as indicated by 95% confidence intervals. Initial differences in relative species abundance and dominance endured, if not increased, over the 100-year simulation period.

Thinned areas initially had more trees, but since untreated areas had more regeneration, they quickly became denser; this difference slowly declined over the course of the simulation (Figure 4.4a). All treatment/regeneration combinations led to some self-thinning, but Regen-2 (scheduling measured regeneration in 2004 and adjusted regeneration in 2024) in untreated areas led to an especially high density and a correspondingly steep decline. After 100 years, thinned and untreated areas were nearly identical, but density under Regen-2 remained higher than that under Regen-1 (scheduling measured regeneration only in 2004). Confidence intervals overlapped for the entire simulation, indicating high plot-to-plot variability. Thinned areas had significantly greater basal area for at least four decades (Figure 4.4b). After 100 years, basal area was very similar under all treatment/regeneration combinations.

Trees in untreated areas were larger on average immediately post-fire due to high mortality of small trees compared to thinned areas (Figure 4.4c). Because of slightly higher regeneration in untreated areas, average tree diameter soon dropped below that of thinned areas. As with tree density, this difference lessened over time, and by the end of the simulation thinned and untreated areas were nearly the same, with a slightly larger average diameter under Regen-1.

In both thinned and untreated areas, an open forest with some surviving ponderosa pines and a few survivors of other species quickly gives way to a thicket of oaks, junipers, and New Mexico locust, with the occasional remnant ponderosa pine (Figure 4.5). Ponderosa pines become a much smaller component of the ecosystem in untreated areas with regard to density (Figures 4.5a-b). Ponderosa pines constutite about five percent of total density under Regen-1 and 10% under Regen-2. In thinned areas, about a quarter of the trees are ponderosa pines after the 2004 regeneration becomes established, and this proportion remains stable over time (Figures 4.5c-d). Thinned areas also have many more ponderosa pines than untreated areas in absolute numbers. Two of the seven untreated sites had only 21 and 26 ponderosa pines/hectare by the end under Regen-1. The relative density of New Mexico locust declines slowly, from 15-30% at its maximum to 10-18% after 100 years. The proportion of oaks increases slightly over time. About half the trees are oaks in untreated areas, and 35% in thinned areas. The proportion of junipers is roughly the same in all treatment/regeneration combinations (~25%) and remains steady over the course of the simulation.

The difference in basal area between thinned and untreated areas increases over time, and there is very little difference between the regeneration scenarios (Figure 4.6).

Ponderosa pine initially makes up all the basal area in untreated areas, but after four decades, thinned areas have proportionally more ponderosa pine. After 100 years, thinned areas are about 60% ponderosa pine, while untreated areas are down to 35%. There appears to be a reciprocal relationship between oaks and pine; given this difference in ponderosa pine basal area between treatments, oaks are largely responsible for the similarity in total basal area between thinned and untreated areas by the end of the simulation. In thinned areas, oaks originally constitute about five percent and are never higher than 20% of the total basal area, whereas in untreated areas they comprise 40% of the total basal area after 100 years.

Discussion

Despite displaying some of the most extreme fire behavior ever observed in the Southwest (USFS 2002), the Rodeo-Chediski fire still decreased in severity in areas that underwent fuel reduction treatments within approximately twelve years before the fire. Thinned areas were significantly different from untreated areas in post-fire forest structure, including density, basal area, and diameter distribution. Thinned areas were also significantly different from untreated areas in some attributes of initial post-fire recovery, including fuel loadings and manzanita density. We expect future forest development to differ as well, especially in terms of dominance by ponderosa pine.

Sampling two years post-wildfire allows us to estimate future forest development more accurately. McHugh and Kolb (2003) ascertained that most mortality after wildfire in northern Arizona ponderosa pine forests occurs within the first two years, so our measurements of survival should be reliable. Regeneration was well-established after the passage of two growing seasons, at least for sprouting species, so our projections of future growth are based on more complete data than could be gathered very soon after the fire.

Because we used variable-radius plots, we do not have a complete sample of trees that had already fallen after being killed by the fire. Fallen trees were included in fuel loadings, since they could be recorded on the planar fuel transect. Our calculations of pre-fire forest structure are therefore an underestimate. However, because we rarely encountered already-fallen trees in the study area, we believe our estimates to be largely accurate.

Pre-wildfire treatment effects

Thinned areas experienced much less extreme fire behavior and effects than untreated areas. This is in accordance with previous studies indicating the effectiveness of fuel reduction treatments at reducing wildfire severity (Martinson and Omi 2003, Finney et al., in press). Overstory survival was far higher in thinned areas, where half the trees survived compared to five percent in untreated areas. After the fire, basal area and density of live trees was higher in thinned areas, and the diameter distribution did not substantially change shape in thinned areas as compared to untreated areas. Some portions of thinned as well as untreated areas still experienced complete overstory mortality, but this was limited to 11% of thinned plots as opposed to 66% of untreated plots. The fire raised crown base height in untreated areas, but not in most portions of thinned areas. While the characteristics we tested did not indicate that pre-fire forest structure on thinned and untreated sites was significantly different, average overall tree density and density of small trees on untreated sites was higher than on thinned sites.

When compared to the treatments tested by Pollet and Omi (2002), our estimate of differences in pre-fire forest structure between thinned and untreated areas is within the range of treatments they also found effective. Our untreated sites were 1.7 times denser than our thinned sites; the untreated sites they studied had between 1.4 and 8.7 times more trees than their treated sites. Pre-fire total basal area was nearly identical on our thinned and unthinned sites, which Pollet and Omi (2002) also found to be the case on two of the four wildfires they sampled. Finney et al. (in press) obtained similar results for treatment effect on burn severity via an analysis of treatment areas and satellite-measured burn severity (ΔNBR). While they only investigated the effect of prescribed burning treatments on the burn severity of the Rodeo-Chediski fire, it should be noted that they did not exclude areas which had also received silvicultural treatments such as pre-commercial thinning from their analysis.

Future forest development

Regeneration two years after the fire was strongly dominated by Gambel oak, alligator juniper, and New Mexico locust, making it very different from the ponderosa pine-dominated, pre-fire and post-fire overstory. Overall regeneration levels were slightly, but not significantly, higher on untreated sites. Ponderosa regeneration was four times higher in thinned areas, but it was very patchily distributed; ponderosa regeneration was found on only six percent of all plots. Manzanita and Fendler's ceanothus were common across all study sites, but manzanita was twenty times more abundant on untreated sites, to the point that it may be inhibiting regeneration.

Our modeling results indicate that thinned areas may have fewer and larger trees and higher basal area than untreated areas for the next several decades. Both thinned and untreated sites became a thicket of young oaks, junipers, and New Mexico locust, with both remnant and young ponderosa pines more common on thinned sites but still not comprising more than a quarter of all trees. Despite no significant difference in regeneration measured in 2004 between thinned and untreated sites, the slightly higher average regeneration levels on untreated sites may have led to their greater tree density over the course of the simulation, especially under the second regeneration scenario. Ponderosa pines still constituted the majority of basal area in thinned areas after 100 years, but accounted for only 35% of total basal area in untreated areas. The proportion of oaks in both density and basal area increased over time in thinned and untreated areas.

Savage and Mast (2005) delineated several trajectories southwestern ponderosa pine forests have taken after crownfire. These included a return to a dense pine forest, recovery to an early-successional ponderosa pine/Gambel oak forest, or a type conversion into a grassland or oak/manzanita shrubfield. Moir and Dieterich (1988) proposed a similar alternative outcome of the successional process for ponderosa pine ecosystems, in that meadows as well as open old-growth forests might be perpetuated by frequent low-intensity fire. There is also preliminary evidence of a post-crownfire transition of a pine/oak forest in southeastern Arizona to an oak woodland (Barton 2002). Since some regeneration was present on every site and 86% of the plots, conversion to grassland is unlikely. Sixty-four percent of untreated and 11% of thinned plots had no surviving pines and no ponderosa regeneration; conversion to an oak/manzanita shrubfield may occur in these areas. In both thinned and untreated areas, future growth will in general be a more balanced combination of oaks, junipers, pines, and New Mexico locust, rather than the strongly ponderosa-dominated, pre-wildfire forest.

Climate change is projected to accelerate shifts in vegetational states caused by fire. Average temperature and precipitation are both expected to increase in the western U.S. and Canada as a result of global climate change. This change is expected to extend the fire season, increasing both fire frequency and total area burned because of the increase in amplitude and duration of extreme fire weather. In the Southwest, large fires are sometimes associated with a current-year drought after several years of above-average precipitation, so higher average precipitation may increase the number of very large fires (Swetnam and Betancourt 1990, Grissino-Mayer and Swetnam 2000, McKenzie et al. 2004). This effect of climate change on fire regimes may indirectly hasten vegetation shifts more than the direct effect of climate change itself on vegetation. Burn areas offer sites for migrating species to colonize, and increased fire size and frequency will decrease the habitat available to late-successional species and increase fragmentation (Flannigan et al. 2001, McKenzie et al. 2004).

It is uncertain how long an altered successional trajectory such as an oak/manzanita shrubfield may persist, and what stage might occur next. Savage and Mast (2005) found that these conditions could persist for at least five decades, and pointed out that these type conversions may be self-perpetuating if the area repeatedly experiences intense fire. On the other hand, a shrubfield near an area with surviving pines may eventually revert to a ponderosa pine forest. Our results indicate that this may be the case for most of the burn area; while 38% of the plots overall had no surviving ponderosa pine trees and no ponderosa pine regeneration, this was not the case for any of the study sites (10 plots per study site), so shrubfields will probably be patchily distributed. Even if these areas ultimately become ponderosa pine forests again, the recovery process will most likely take centuries rather than decades. Savage and Mast (2005) observed this patchy shrubfield/forest state fifty years after the 1950 Faught Ridge fire, very close to the Rodeo-Chediski burn area. During the interim, ponderosa pine forests on the burn area will be fragmented and considerably reduced in total size.

Management implications

The Rodeo-Chediski fire was not as completely disastrous as is often believed, largely because of the fuel reduction treatments that took place before the fire. These treatments comprised only a small portion of the burn area, however.

Ponderosa pine will lose dominance for the next several decades to centuries over a considerable proportion of the burn area, to be replaced by a matrix of oak/manzanita shrubfields and thickets of oaks, junipers, and New Mexico locust with the occasional remnant ponderosa pine. Most thinned areas should recover to a ponderosa pine/Gambel oak forest relatively quickly, but because of sparse ponderosa regeneration, it may be necessary to do plantings, especially in untreated areas.

Most southwestern forests are currently at high risk of severe crownfire; unless extensive fuel reduction treatments are undertaken in the near future, these forests will experience intense fires after which recovery will be extremely slow. The completely unintended effect of fire suppression, overgrazing, and conventional timber management focusing on the harvesting of large trees may be the increasing fragmentation of the largest contiguous ponderosa pine forest in the United States by landscape-scale fires such as the Rodeo-Chediski, with resultant long-term soil damage (Viro 1974), loss of timber revenue, and ecological and social effects that are not yet entirely known. Climate change is forecasted to accelerate these shifts (McKenzie et al. 2004).

In 1951, Weaver recommended thinning of small-diameter trees and the use of prescribed fire to improve forest health and prevent catastrophic wildfire in ponderosa pine forests. Forest management involving fuel reduction treatments now also recommended by many others (Agee and Skinner 2005), such as regular pre-commercial thinning including slash treatment, prescribed burning, and the combination of thinning and prescribed burning, should provide the best protection against severe crownfire and ensuing conversion away from ponderosa-dominated forests. Restoration to presettlement conditions (Covington et al. 2001) may also offer additional benefits for native species. While fuel reduction treatments within the wildland/urban interface should continue to be our first priority, we ought not to allow more distant forests to remain in jeopardy.

Table 4.1. Overstory and regeneration species found on the Apache-Sitgreaves National Forest post-wildfire, all conditions. "% Presence" indicates the percent of plots on which the species was found. “Overstory” indicates a tree at least 1.4 m tall, and “Regeneration” indicates tree regeneration under 1.4 m.

Common name	**Scientific name**	**% Presence (Overstory)**	**% Presence (Regeneration)**
Ponderosa pine	*Pinus ponderosa*	99	6
Alligator juniper	*Juniperus deppeana*	35	31
Gambel oak	*Quercus gambelii*	34	65
Utah juniper	*Juniperus osteosperma*	1	0
Scrub (turbinella) oak	*Quercus turbinella*	1	4
Chihuahua pine	*Pinus leiophylla*	1	0
Southwestern white pine	*Pinus strobiformis*	1	0
Other oaks (unidentified)	*Quercus* spp.	0	6
New Mexico locust	*Robinia neomexicana*	0	4

Table 4.2. Shrub species found on the Apache-Sitgreaves National Forest, all conditions. "% Presence" indicates the percent of plots on which the species was found.

Commmon name	Scientific name	% Presence
Fendler's ceanothus	*Ceanothus fendleri*	19
Pinemat manzanita	*Arctostaphylos pungens*	13
Pringle manzanita	*Arctostaphylos pringlei*	1
Mountain mahogany	*Cercocarpus montanus*	1

Table 4.3. DISTLM (Anderson 2004) was used to make univariate and multivariate comparisons across treatments. The Bray-Curtis dissimilarity distance measure was used, and the number of permutations for each test was 999. P-values and R^2 (proportion of variation explained) are below; non-significant *p*-values ($p > 0.05$) are italicized.

Characteristic	***P*-value**	**R^2**
Trees/hectare (live)	0.001	0.22
Trees/hectare (live+dead)	*0.966*	< 0.01
Survival (based on trees/hectare)	0.001	0.26
Basal area (live)	0.001	0.26
Basal area (live+dead)	*0.781*	< 0.01
Crown base and bole char height, live	0.005	0.07
Bole char height, all	0.001	0.23
Diameter distribution (live)	0.001	0.18
Diameter distribution (live+dead)	*0.856*	< 0.01
Snags	0.001	0.18
Fuels, grouped by forest floor/fine/coarse	0.001	0.05
Regeneration, all (unstandardized)	*0.097*	0.01
Regeneration, all (standardized by species)	*0.083*	0.01
Pinus ponderosa	*0.078*	0.02
Quercus gambelli	*0.128*	0.01
Shrubs, all (unstandardized)	*0.250*	0.01
Shrubs, all (standardized by species)	*0.359*	0.01
Arctostaphylos pungens	0.043	0.02

Figure 4.1. Burn severity over the extent of the Rodeo-Chediski fire was mixed. Seven pairs of thinned and unthinned sites were sampled on the northern portion of the burn area within the Apache-Sitgreaves National Forest. The burn area and the Apache-Sitgreaves National Forest in relation to the state of Arizona are shown in the lower left.

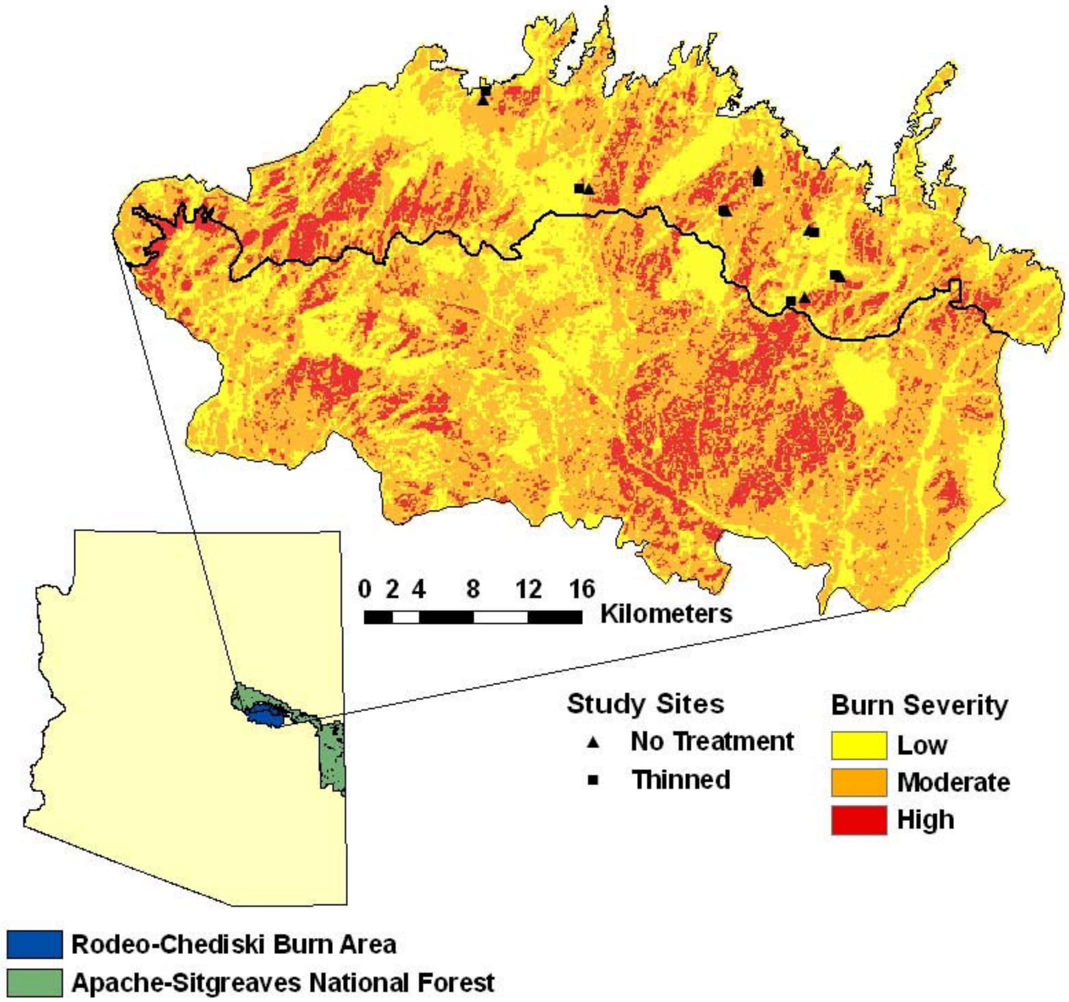


Figure 4.2a. Tree density was not significantly different pre-wildfire (p = 0.966), but was much higher in thinned areas than in untreated areas post-fire (p = 0.001). Survival was about 50% in thinned areas, compared to 5% in untreated areas. Error bars for this and subsequent figures represent ±1 standard error unless otherwise noted.

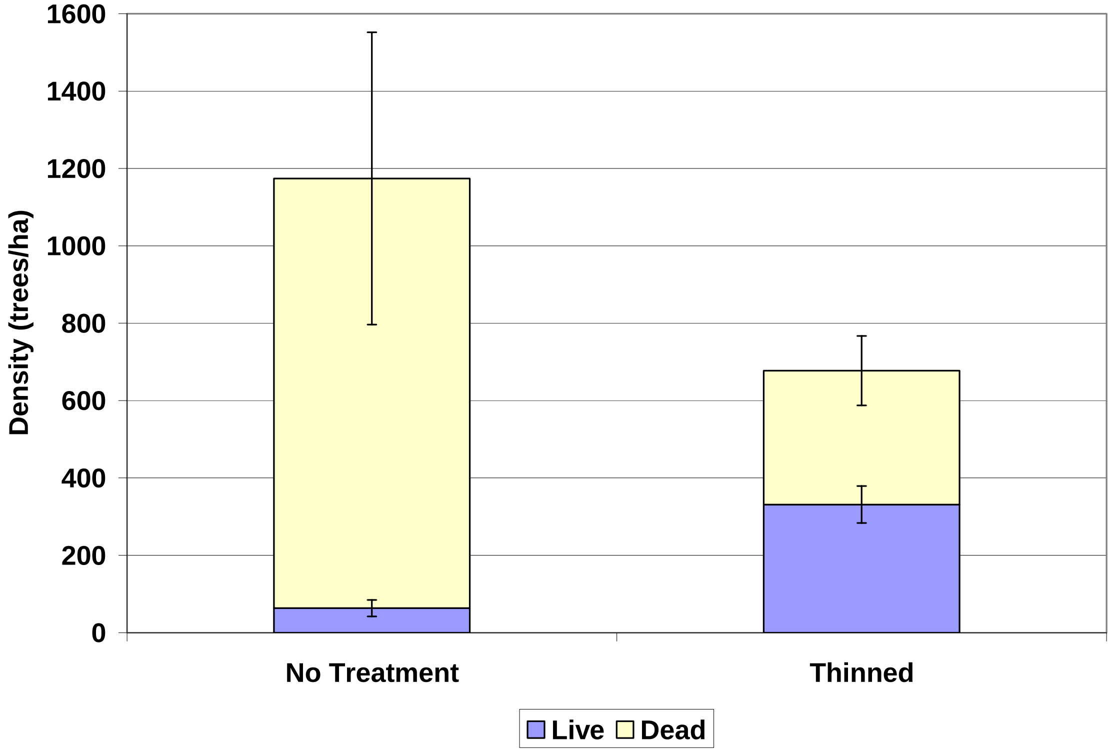


Figure 4.2b. Live tree basal area was similar before the fire ($p = 0.781$), but was much higher in thinned areas after the fire ($p = 0.001$).

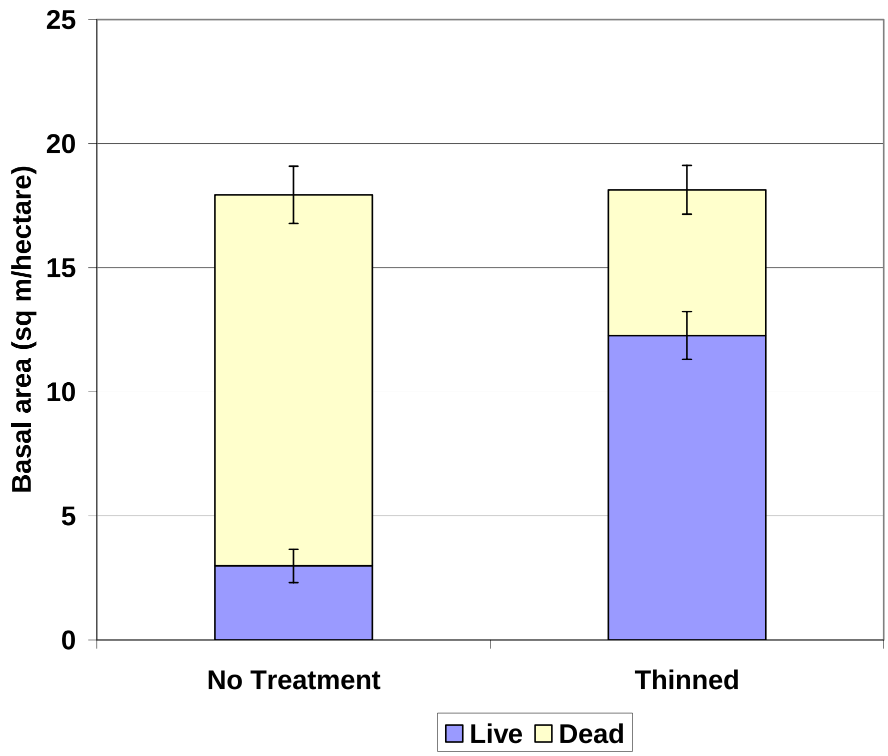


Figure 4.2c. Crown base and bole char height for live trees (p = 0.005) and bole char height for all trees (p = 0.001) was lower in thinned areas.

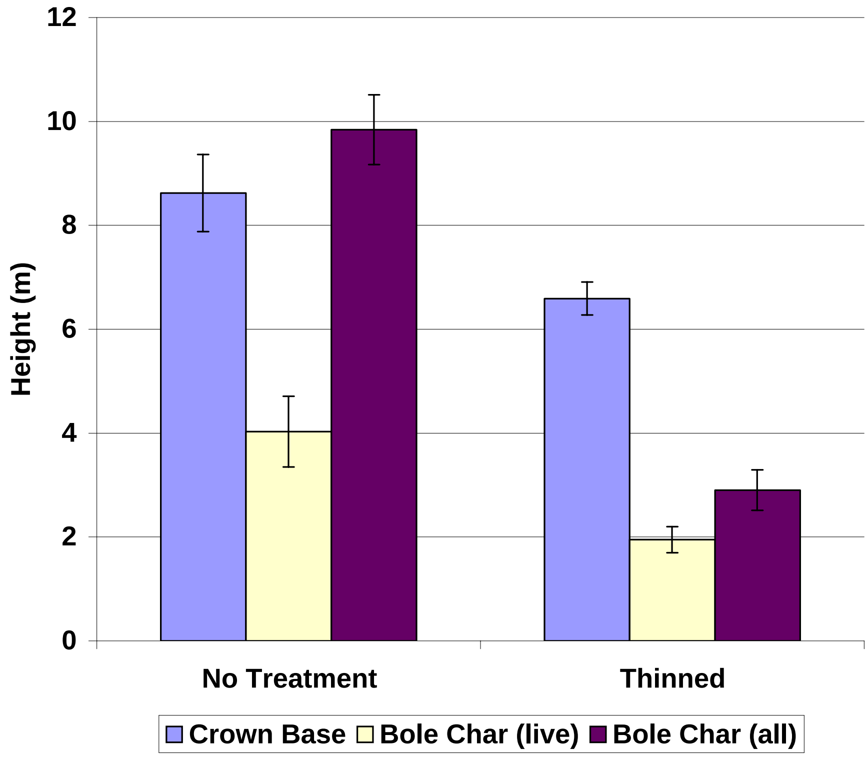


Figure 4.2d. The diameter distribution for ponderosa pine was not significantly different in thinned and untreated areas before the fire ($p = 0.856$), but was different after the fire ($p = 0.001$). The fire shifted the diameter distribution for ponderosa pine in favor of larger trees in untreated areas, but did not greatly change it in thinned areas. Diameter classes are in centimeters.

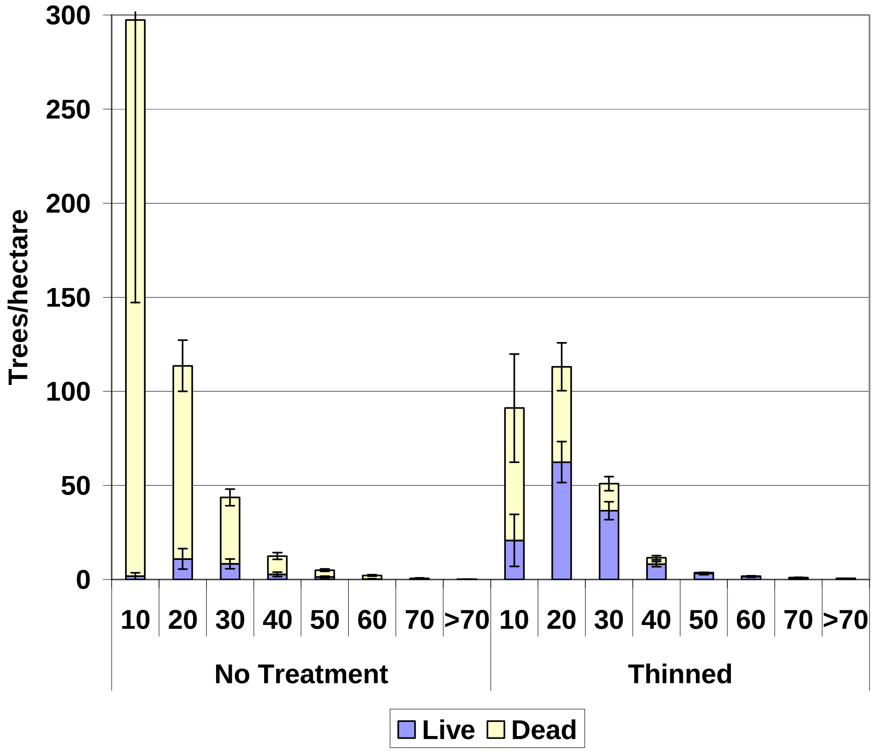


Figure 4.2e. Cumulative age frequency for all species. About 95% of trees were less than 100 years old in both thinned and untreated areas.

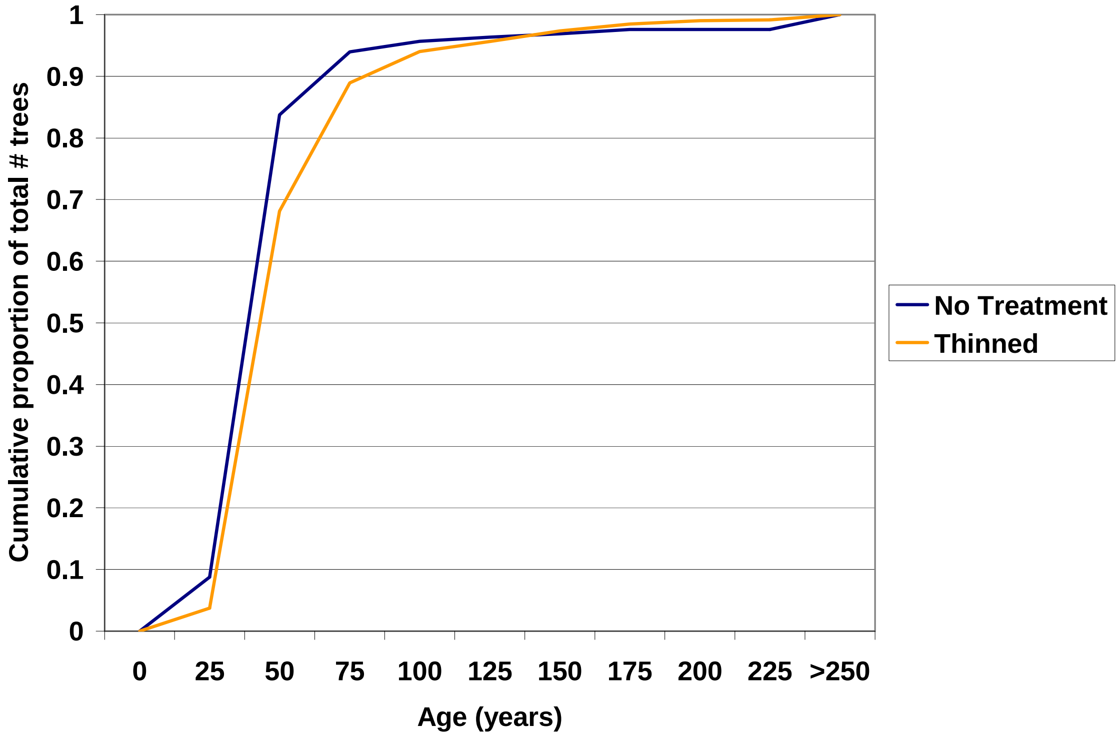


Figure 4.2f. There were many more standing snags of two common size classes used by wildlife (>30 cm and >50 cm DBH) in untreated areas (p = 0.001).

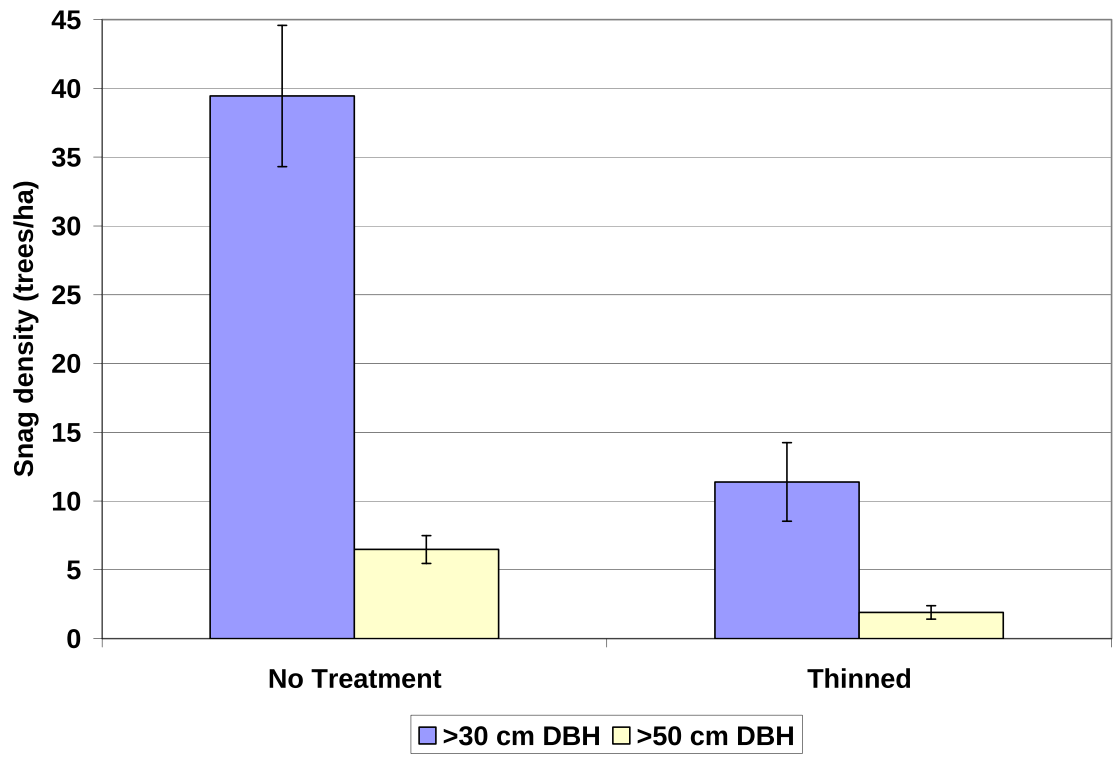


Figure 4.3a. Post-fire fuel loadings were higher in thinned areas (p = 0.001).

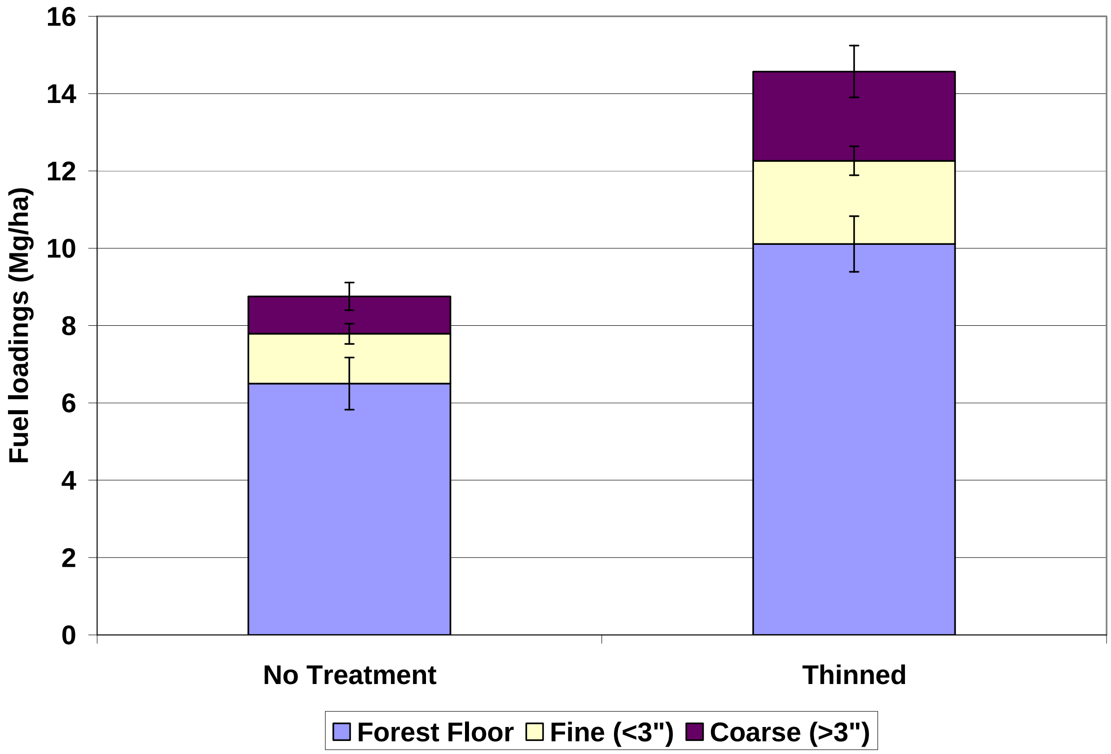
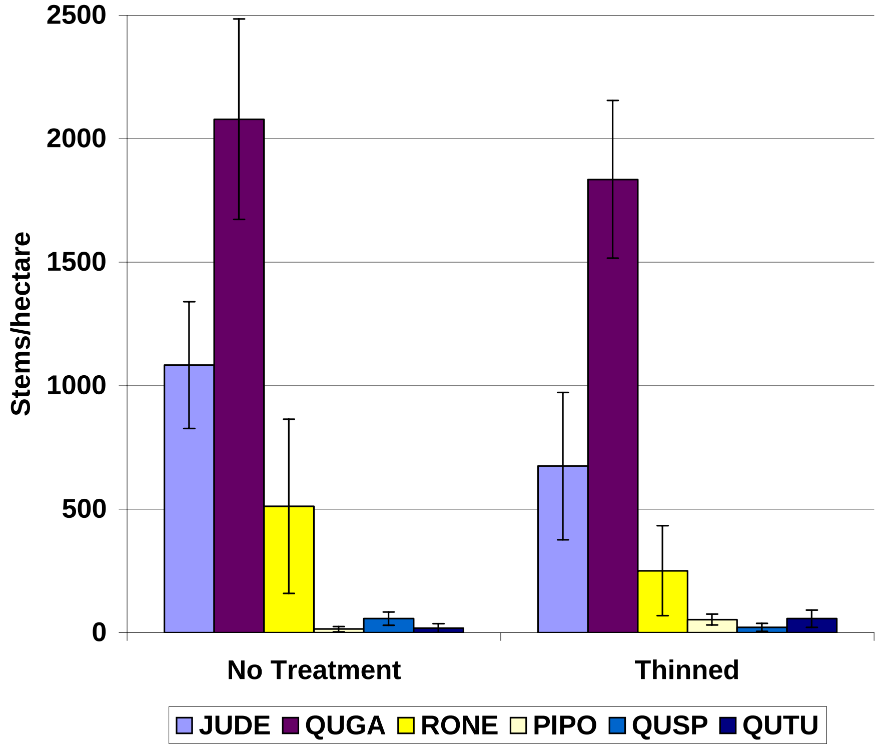


Figure 4.3b. Regeneration was dominated by sprouting species such as junipers and oaks, as well as by New Mexico locust. While untreated areas had slightly more regeneration on average, overall regeneration was not significantly different on thinned vs. untreated areas ($p = 0.097$).



JUDE = *Juniperus deppeana*, QUGA = *Quercus gambelii*, RONE = *Robinia neomexicana*, PIPO = *Pinus ponderosa*, QUSP = Other *Quercus* species (grouped), QUTU = *Quercus turbinella*.

Figure 4.3c. Ponderosa pine regeneration, broken down by height class. While ponderosa pine regeneration was four times higher on average in thinned areas than untreated areas, this was not a significant difference ($p = 0.078$).

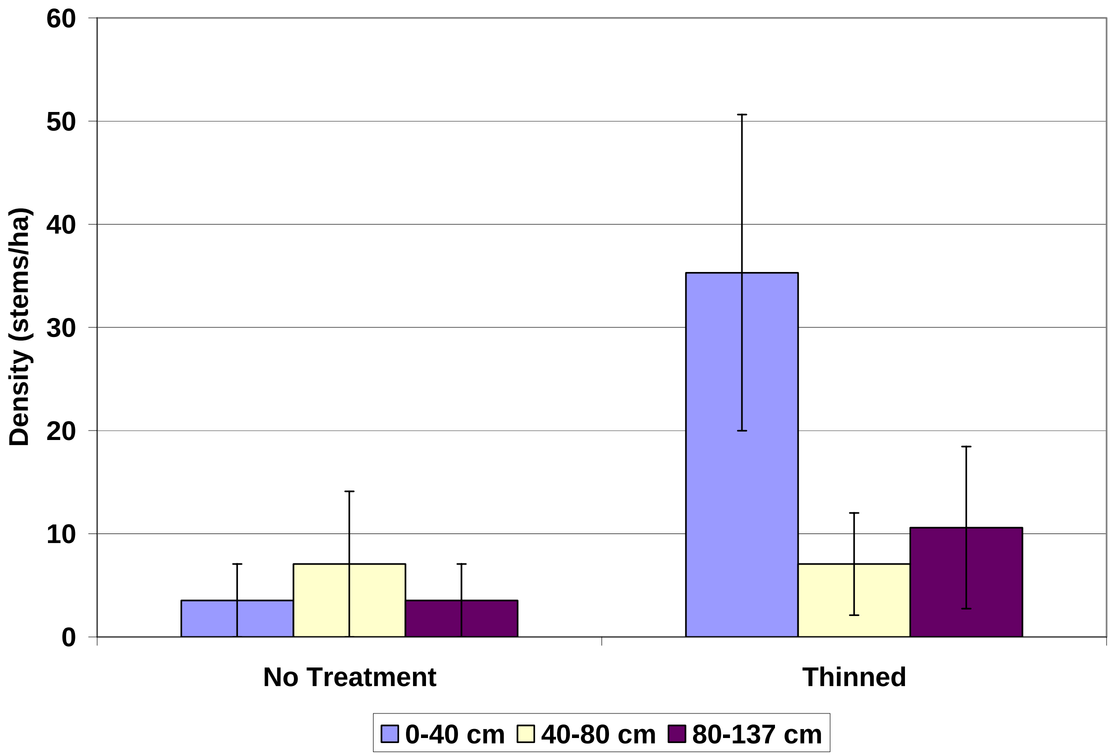
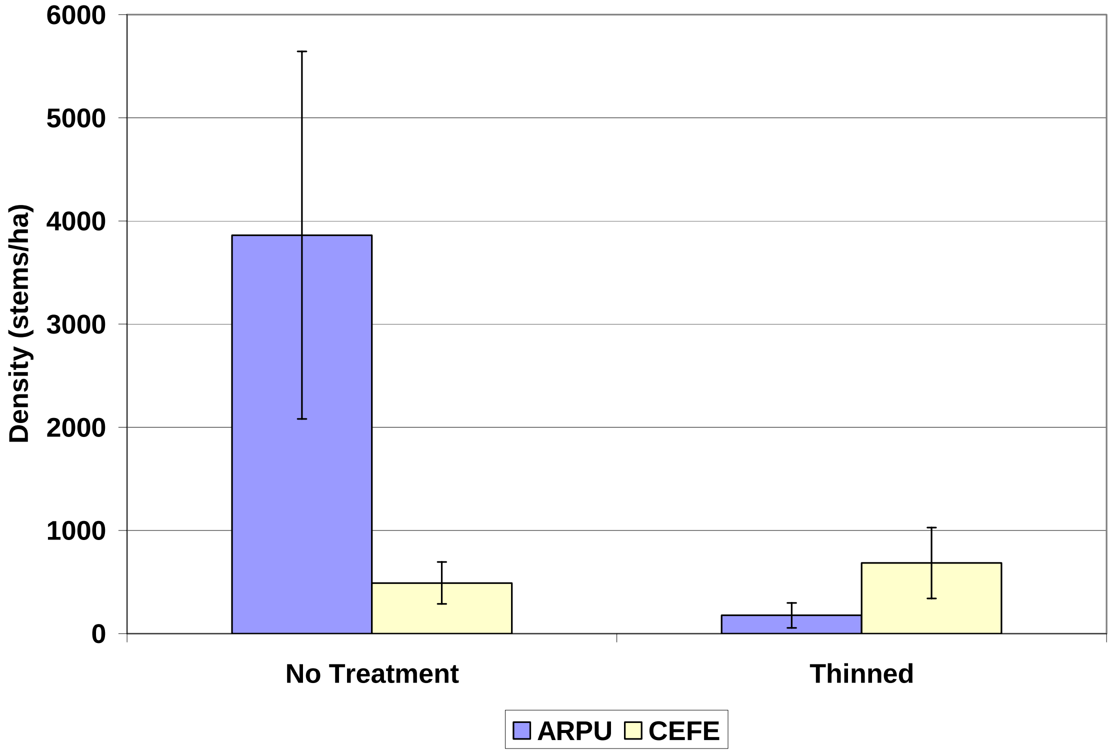


Figure 4.3d. Manzanita and Fendler's ceanothus were very common. Untreated areas had twenty times as much manzanita as thinned areas (p = 0.043).



ARPU = *Arctostaphylos pungens*, CEFE = *Ceanothus fendleri*.

Figure 4.4a. Projections of future forest growth using the Forest Vegetation Simulator (Dixon 2003) indicate that untreated areas quickly become denser than thinned areas due to more regeneration (all species included). Self-thinning occurs under all conditions and density converges within 100 years. NT = No Treatment; TH = Thinned; R1 = Regeneration Scenario 1; R2 = Regeneration Scenario 2. Error bars represent 95% confidence intervals for this and subsequent figures.

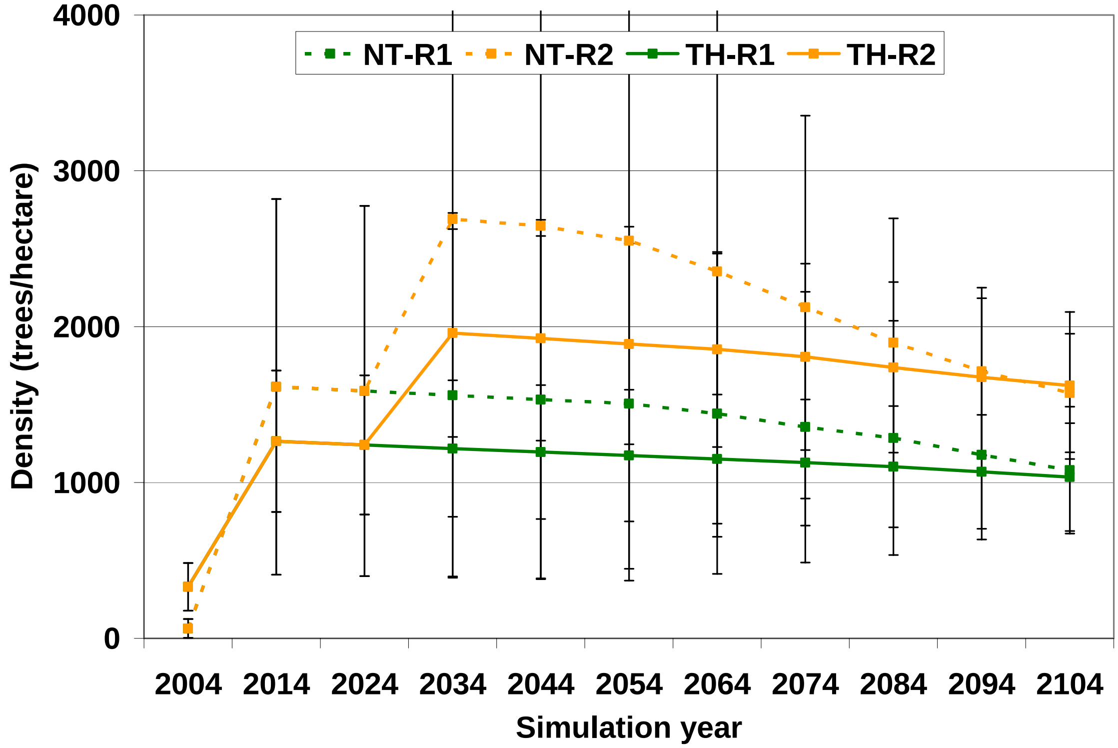


Figure 4.4b. Total basal area for all species remains higher in thinned areas for several decades. The regeneration scenarios make little difference in basal area.

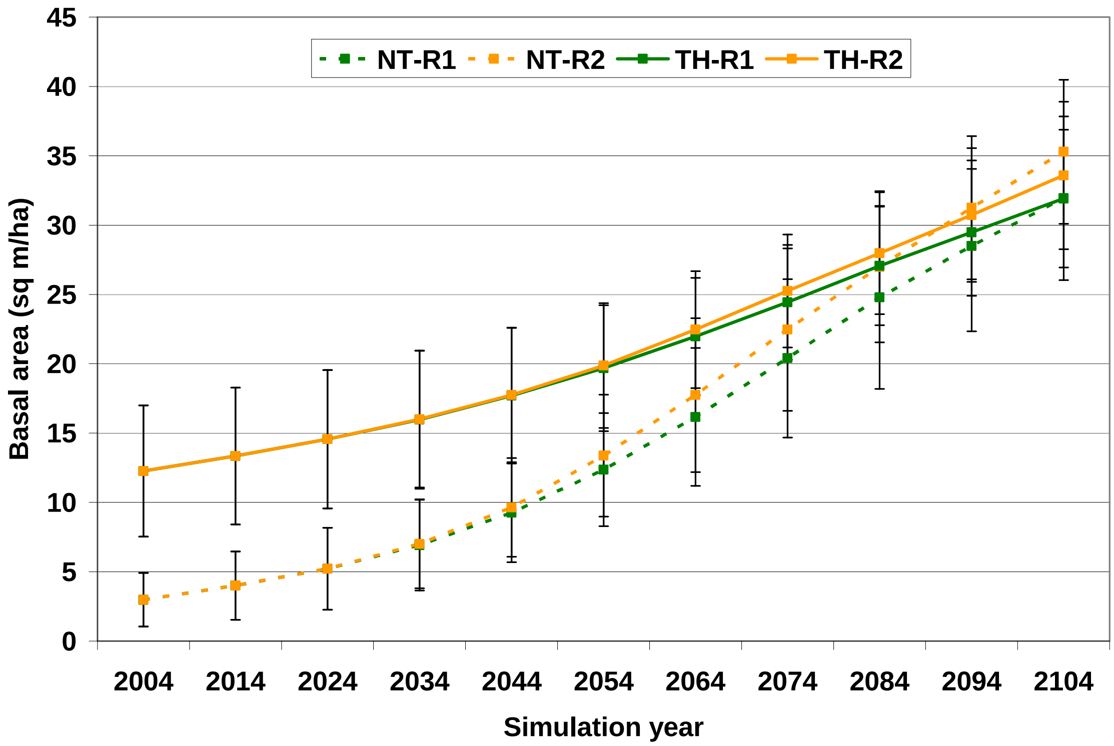
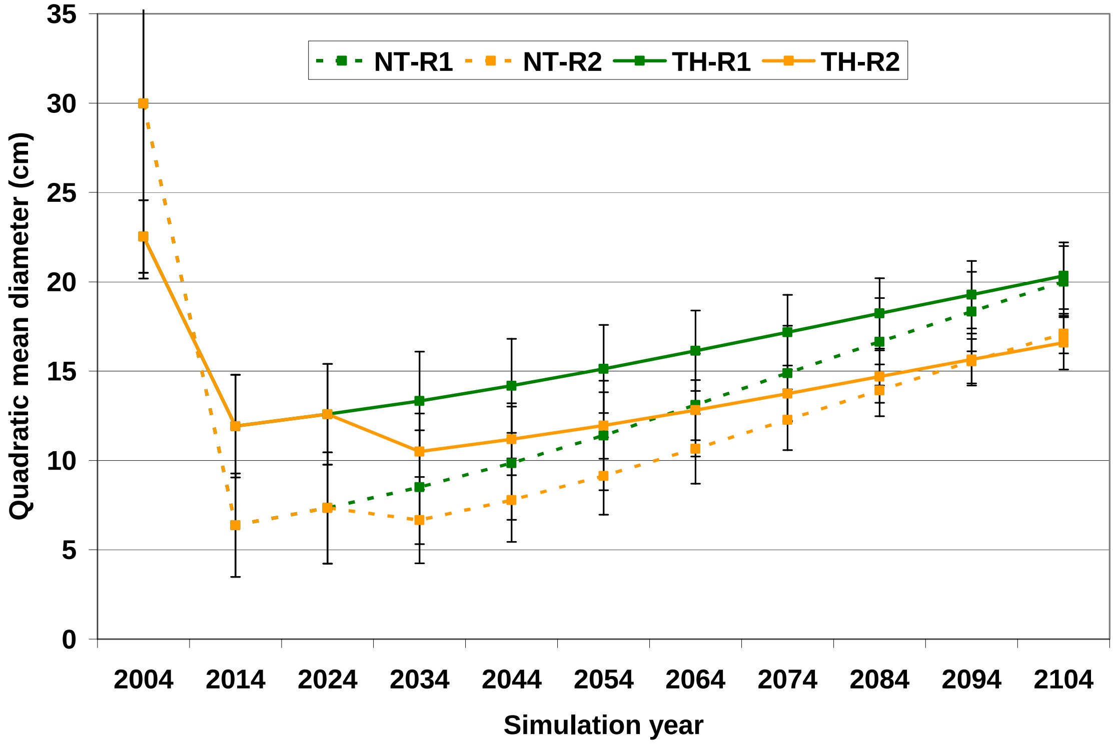


Figure 4.4c. Quadratic mean diameter is initially higher in untreated areas, then drops sharply due to the larger number of small trees. QMD recovers to that of thinned areas immediately post-fire on both thinned and untreated areas by the end of the simulation.



a. b.

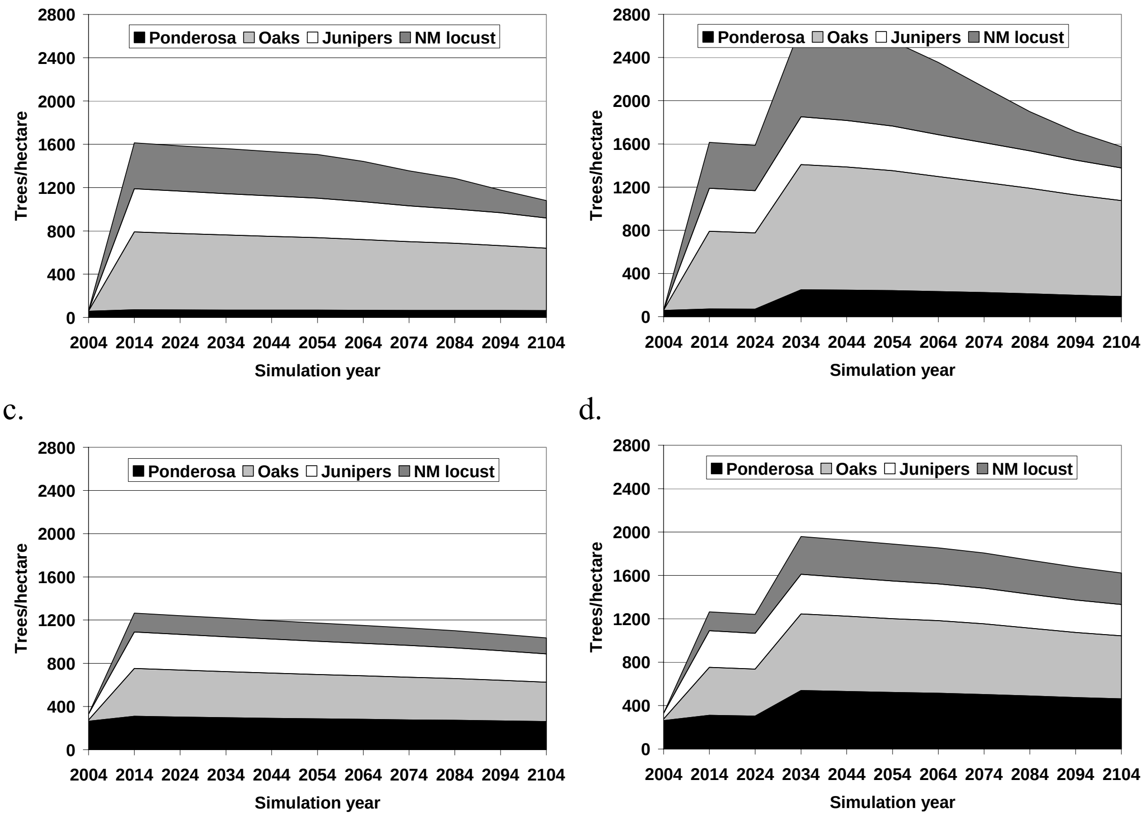


Figure 4.5. Projected tree density by species.

a. No treatment, Regeneration Scenario 1.
b. No treatment, Regeneration Scenario 2.
c. Thinned, Regeneration Scenario 1.
d. Thinned, Regeneration Scenario 2.

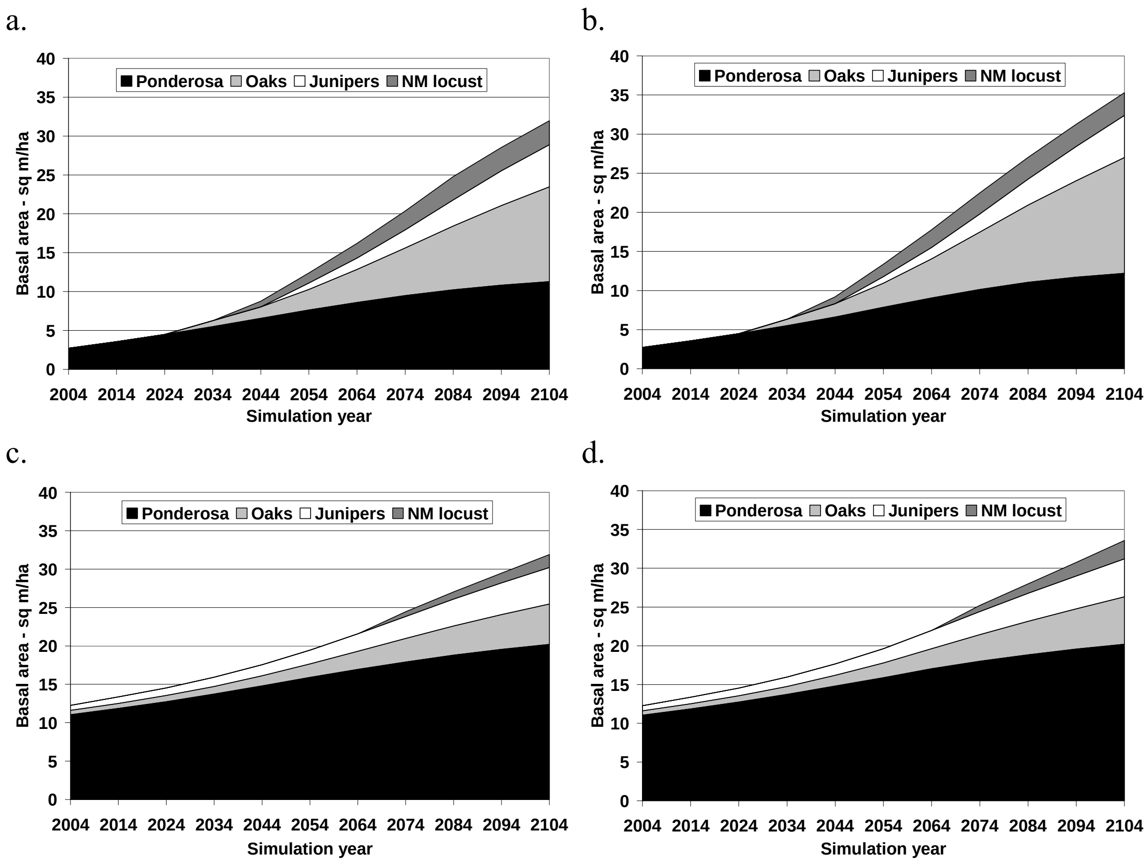


Figure 4.6. Projected basal area by species.

a. No treatment, Regeneration Scenario 1.
b. No treatment, Regeneration Scenario 2.
c. Thinned, Regeneration Scenario 1.
d. Thinned, Regeneration Scenario 2.

5. DISCUSSION

The individual manuscript chapters (3 and 4) included discussions of the results and management implications of each study. This final chapter presents comparisons between the study areas on the Apache-Sitgreaves National Forest and White Mountain Apache Tribal lands, overall management implications, and future directions for research.

I. Differences and Parallels

Pre-fire fuel reduction treatments significantly reduced burn severity on both the Apache-Sitgreaves National Forest (A-S) and the White Mountain Apache Tribal lands (WMAT). The combination of timber harvesting including pre-commercial thinning and prescribed burning had the greatest ameliorative effect on burn severity. Prescribed burning alone also considerably reduced burn severity, provided that it took place within the decade before the fire. This is in accordance with previous studies indicating the effectiveness of fuel reduction treatments at reducing wildfire severity, including the effectiveness of recent prescribed burning alone (Martinson and Omi 2003).

Treated areas had more live trees, in terms of percent survival (calculated for the A-S only), density, and basal area. These areas also experienced less extreme fire behavior, as indicated by bole char height. After the wildfire, there were more live trees overall on the WMAT than on the A-S in untreated areas, but similar numbers in treated areas. Untreated areas on the A-S had 63 trees/hectare, as opposed to 170 trees/hectare on the untreated portions of the WMAT (averaging across all severities). Thinned areas on the A-S had 331 trees/hectare, comparable to the 396 trees/hectare in burn only areas and 305 trees/hectare in the cut and burned areas of the WMAT. The difference was strongest in terms of smaller trees, which may reflect the effects of BDq-based uneven-aged management on the WMAT as compared to the even-aged management in common use on the A-S.

The WMAT study area was more diverse than the A-S; we found twenty tree and eleven shrub species on the WMAT, compared to nine tree and four shrub species on the A-S. This may be partially explained by the larger number of plots sampled on the WMAT (444 plots vs. 140 plots on the A-S), but is probably still a meaningful difference given the greater topographical variation of the WMAT study area and its wider elevational range. The A-S study sites ranged in elevation from 1990 – 2138 m, while WMAT study sites ranged from 2000 – 2293 m. Thus, our WMAT results may be applicable to a wider variety of ponderosa pine habitat types.

Regeneration across both study areas was dominated by sprouting species such as oaks and junipers, as well as by New Mexico locust. Ponderosa pine regeneration was sparse to nonexistent in untreated areas, common in treated areas, and patchily distributed in general. Untreated areas of the A-S and WMAT had similar amounts of ponderosa regeneration, but treated areas of the WMAT had much more than the A-S. Untreated portions of the A-S had 14 stems/hectare, comparable to 21 stems/hectare on the WMAT. Thinned areas of the A-S had 53 stems/hectare, while the treated areas of the WMAT had nearly an order of magnitude more: 228 stems/hectare in burn only areas, and 457 stems/hectare in cut and burned areas. This result is probably due to the larger number of trees surviving on the WMAT to act as seed sources.

Fendler's ceanothus (*Ceanothus fendleri*) was common in both study areas but, like ponderosa pine regeneration, was nearly an order of magnitude more abundant on the WMAT. Untreated and thinned areas of the A-S had approximately 500 and 700 stems/hectare, respectively. Untreated, burn only, and cut and burned areas of the WMAT had twelve, twenty, and nine thousand stems/hectare, respectively. Fendler's ceanothus appeared to respond strongly to prescribed burning on the WMAT; along with other inherent differences between the study areas, the more widespread use of prescribed burning on the WMAT may explain its greater abundance there.

Pinemat manzanita (*Arctostaphylos pungens*) was also common in both study areas. It was more common in untreated areas on both the A-S and the WMAT, and more abundant in treated areas of the WMAT than of the A-S. It was twenty times more abundant on untreated A-S sites (3900 stems/ha) than on treated sites (175 stems/ha), to the point that it may be competing with tree regeneration. Its distribution was more uniform on the WMAT. There were 840 stems/hectare in untreated areas, and about 320 stems/hectare in both burn only and cut and burned areas. Because the environment of the WMAT is more diverse in terms of elevation and topography, the greater variation in manzanita abundance on the A-S might be due to another environmental factor such as soils, or to less uniformity in the treatment history.

A detailed comparison of forest recovery for the two studies is difficult, because we do not have the spatial data necessary to expand the A-S results to the landscape level, did not design the A-S study to compare treatments across a range of burn severities, and have not yet modeled future growth on the WMAT. Both indicate that portions of the study area with few surviving ponderosa pine trees and little to no ponderosa pine regeneration may transition to an oak/manzanita shrubfield, in accordance with the findings of Savage and Mast (2005). Ponderosa pine may lose dominance in untreated areas of the WMAT and A-S, perhaps for well over a century. Because of the substantial difference in ponderosa pine regeneration between the WMAT and the A-S, it is somewhat doubtful that the WMAT findings can appropriately be extrapolated to the A-S. However, they do indicate that higher-severity thinned areas of the A-S may have sufficient regeneration for a ponderosa pine forest to become reestablished even with low numbers of mature ponderosas.

II. Ecosystem management implications and future directions

If the trend of increasing size and frequency of crownfires in ponderosa pine forests continues, as is likely (Agee and Skinner 2005), much of the Southwest may soon resemble the Rodeo-Chediski burn area. Our results strongly indicate that although the fire was extremely severe, areas with fuel reduction treatments had considerably lower burn severity and should recover to ponderosa pine-dominated forests relatively soon. Some portions of untreated areas may transition to self-perpetuating shrubfields, an unintended consequence with social, economic, and ecological effects that are not yet fully known.

The period before the Rodeo-Chediski fire during which the fuel reduction treatments we studied took place was similar for both studies: 1991 and later for the White Mountain Apache Tribal lands, and 1990 and later for the Apache-Sitgreaves National Forest. This treatment period of 10-11 years was effective at reducing burn severity. These fuel reduction treatments generally did not stop the advancement of the fire, but some treated areas did appear to did slow the fire's advancement, change its direction, and protect some adjacent untreated areas (Finney et al., in press).

Fuel reduction treatments can change fire behavior and effects even under the most extreme conditions. The WMAT study indicated that the post-fire differences among treatments became more pronounced as burn severity increased. While fuel reduction treatments may be expensive, especially at the frequency that our results demonstrated may be necessary, they have a substantial effect on the sustainability of ponderosa pine forests, and are far cheaper than fire suppression and rehabilitation.

We plan to model future forest development on the White Mountain Apache Tribal lands using FVS, with methods similar to those used for the Apache-Sitgreaves study. While our model results for the A-S study indicated that shifts in species composition may persist for at least 100 years especially in untreated areas, we did not subject the sites to any further treatment or other disturbance during the simulation period. It would be informative to test whether the alternative states potentially resulting from the Rodeo-Chediski fire might in fact be self-perpetuating, as suggested by Savage and Mast (2005), by using a model incorporating stochastic fire regimes such as FIRESUM (Keane et al. 1989, 1990). It would also be instructive to model future growth on the Apache-Sitgreaves National Forest and the White Mountain Apache Tribal lands across a range of climate change scenarios, and assess differences in carbon sequestration across treated and untreated areas.

6. LITERATURE CITED

Agee, J.K. 1998. Fire and pine ecosystems. Pages 193-218 *in* D.M. Richardson, editor. Ecology and biogeography of *Pinus*. Cambridge University Press: Cambridge, UK.

Agee, J.K., Bahro, B., Finney, M.A., Omi, P., Sapsis, D.B., Skinner, C.N., van Wagtendonk, J.W., and Weatherspoon, C.P. 2000. The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management* 127:55-66.

Agee, J.K., and Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*: in press.

Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. *Austral Ecology* 26:32-46.

Anderson, M.J., and Robinson, J. 2001. Permutation tests for linear models. *Australian and New Zealand Journal of Statistics* 43:75-88.

Anderson, M.J. 2004. DISTLM v.5: a FORTRAN computer program to calculate a distance-based multivariate analysis for a linear model. Department of Statistics, University of Auckland, New Zealand.

Applequist, M.B. 1958. A simple pith locator for use with off-center increment cores. *Journal of Forestry* 56:141.

Baisan, C.H. and Swetnam, T.W. 1990. Fire history on a desert mountain range: Rincon Mountain Wilderness, Arizona, USA. *Canadian Journal of Forest Research* 20:1559-1569.

Barger, R.L., and Ffolliott, P.F. 1972. Physical characteristics and utilization of major woodland tree species in Arizona. Research Paper RM-83. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 80 pp.

Barton, A.M. 2002. Intense wildfire in southeastern Arizona: transformation of a Madrean oak-pine forest to oak woodland. *Forest Ecology and Management* 165:205-212.

Belsky, A.J. and Blumenthal, D.M. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the interior West. *Conservation Biology* 11:315-327.

Blatner, K.A., and Govett, R.L. 1988. Ponderosa pine lumber market. *In*: Baumgartner, D.M., Lotan, J.E., compilers. Ponderosa pine: the species and its management: Symposium proceedings; 1987 September 29 – October 1; Spokane, WA. Pullman, WA: Washington State University, Cooperative Extension: 7-9.

Brose, P. and Wade, D. 2002. Potential fire behavior in pine flatwood forests following three different fuel reduction techniques. *Forest Ecology and Management* 163:71-84.

Brown, J.K. 1974. Handbook for inventorying downed woody material. General technical report INT-16. Ogden, Utah: USDA Forest Service.

Chapin, FS III; Rupp, TS; Starfield, AM; DeWilde, L-O; Zavaleta, ES; Fresco, N; Henkelman, J; McGuire, AD. 2003. Planning for resilience: modeling change in human-fire interactions in the Alaskan boreal forest. *Frontiers in Ecology and the Environment* 1(5):255-261.

Cocke, A.E. et al. 2005. Fire exclusion and burn severity on the San Francisco Peaks, Arizona. *International Journal of Wildland Fire*; in press.

Cooper, C.F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. *Ecological Monographs* 30:129-164.

Covington, W.W., and Sackett, S.S. 1984. The effect of a prescribed burn in southwestern ponderosa pine on organic matter and nutrients in woody debris and forest floor. *Forest Science* 30(3):183-92.

Covington, W.W., P.Z. Fulé, S.C. Hart, and R.P. Weaver. 2001. Modeling ecological restoration effects on ponderosa pine forest structure. *Restoration Ecology* 9(4):421-431.

Covington, W.W., Fulé, P.Z., Moore, M.M., Hart, S.C., Kolb, T.E., Mast, J.N., Sackett, S.S., and Wagner, M.R. 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. *Journal of Forestry* 95(4):23-29.

Covington, W.W., and M. M. Moore. 1994. Southwestern ponderosa pine forest structure: Changes since EuroAmerican settlement. *Journal of Forestry* 92(1):39-47.

Cram, D.S. and Baker, T.T. 2003. Annual report: Inventory and classification of wildland fire effects in silviculturally treated vs. untreated forest stands of New Mexico and Arizona. College of Agriculture and Home Economics, New Mexico State University. Las Cruces, NM. 46 pp.

Deeming, J.E. 1990. Effects of prescribed fire on wildfire occurrence and severity. In: Natural and prescribed fire in Pacific Northwest forests. Walstad, J. D. et al. (eds). Oregon State Univ. Press, Corvallis, OR. Pp. 95-104.

DeLong, S.C., and Kessler, W.B. 2000. Ecological characteristics of mature forest remnants left by wildfire. *Forest Ecology and Management* 131:93-106.

Edminster, C.B., Mathiasen, R.B., and Olsen, W.K. 1991. GENGYM: a variable density stand table projection system calibrated for mixed conifer and ponderosa pine stands in the Southwest. Research Paper RM-297. Ft. Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 32pp.

Ffolliott, P.F., Clary, W.P., and Davis, J.R. 1968. Some characteristics of the forest floor under ponderosa pine in Arizona. Research note RM-127. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. 7 pp.

Finney, M.A., McHugh, C.W., and Grenfell, I.C. 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. *Canadian Journal of Forest Research*: in press.

Flannigan, M., Campbell, I. Wotton, M., Carcaillet, C., Richard, P., Bergeron, Y. Future fire in Canada's boreal forest: paeoecology results and general circulation model – regional climate model simulations. *Canadian Journal of Forest Research* 31:854-864.

Foxx, T.S. 1996. Vegetation succession after the La Mesa fire at Bandelier National Monument. Gen. Tech. Rep. RM-GTR-286:47-69. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station.

Friederici, Peter (ed.). 2003. Ecological restoration of southwestern ponderosa pine forests. 2003. Island Press, Washington, D.C. 561 pp.

Fulé, P.Z., Covington, W.W., and Moore, M.M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. *Ecological Applications* 7(3):895-908.

Fulé, P.Z., Garcia-Arevalo, A., and Covington, W.W. 2000. Effects of an intense wildfire in a Mexican oak-pine forest. *Forest Science* 46(1):52-61.

Fulé, P.Z., Waltz, A.E.M., Covington, W.W., and Heinlein, T.A. 2001. Measuring forest restoration effectiveness in hazardous fuels reduction. *Journal of Forestry* 99(11):24-29.

Fulé, P.Z., Covington, W.W., Smith, H.B., Springer, J.D., Heinlein, T.A., Huisinga, K.D., Moore, M.M. 2002. Comparing ecological restoration alternatives: Grand Canyon, Arizona. *Forest Ecology and Management* 170:19-41.

Fulé, P.Z., Crouse, J.E., Cocke, A.E., Moore, M.M., and Covington, W.W. 2004. Changes in canopy fuels and potential fire behavior 1880-2040: Grand Canyon, Arizona. *Ecological Modelling* 175:231-248.

Gaines, E.M., Kallander, H.R., Wagner, J.A. 1958. Controlled burning in southwestern ponderosa pine: results from the Blue Mountain plots, Fort Apache Indian Reservation. *Journal of Forestry* 58: 323-327.

Gracia, M., Retana, J., and Roig, P. 2002. Mid-term successional patterns after fire of mixed pine-oak forests in NE Spain. *Acta Oecologica* 23(6):405-411.

Greene, D.F., Noel, J., Bergeron, Y., Rousseau, M., and Gauthier, S. 2004. Recruitment of *Picea mariana*, *Pinus banksiana*, and *Populus tremuloides* across a burn severity gradient following wildfire in the southern boreal forest of Quebec. *Canadian Journal of Forest Research* 34:1845-1857.

Grissino-Mayer, H.D., and Swetnam, T.W. 2002. Century-scale climate forcing of fire regimes in the American Southwest. *Holocene* 10(2):213-220.

He, H.S., Mladenoff, D.J., Gustafson, E.J. 2002. Study of landscape change under forest harvesting and climate warming-induced fire disturbance. *Forest Ecology and Management* 155:257-270.

Hessburg, P.F., Agee, J.K., Franklin, J.F. 2005. Dry forests and wildland fires of the inland northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. *Forest Ecology and Management*: in press.

Institute of Ecosystem Studies. Gap Light Analyzer (computer program). Copyright 1999, Simon Fraser University. http://www.ecostudies.org/gla

Johnson, K.L. 1985. Volume and biomass on Gambel oak woodlands. *In*: Van Hooser, D.D., Van Pelt, N., compilers. Growth and yield and other mensurational tricks: a regional technical conference. Gen. Tech. Rept. INT-193. Ogden, UT: USDA Forest Service, Intermountain Research Station: 82-85.

Kallander, H.R. 1969. Controlled burning on the Fort Apache Indian Reservation, Arizona. Tall Timbers Fire Ecology Conference Proceedings, No. 9:241-250. Tallahassee, FL: Tall Timbers Research Station.

Keane, R.E., Arno, S.F., Brown, J.K. 1989. FIRESUM – An ecological process model for fire succession in western conifer forests. USDA Forest Service General Technical Report INT-266. Intermountain Research Station, Ogden, Utah.

Keane, R.E., Arno, S.F., Brown, J.K. 1990. Simulating cumulative fire effects in ponderosa pine/Douglas-fir forests. *Ecology* 71:189-203.

Keane, R.E., Morgan, P., Running, S.W. 1996. FIRE-BGC – A mechanistic ecological process model for simulating fire succession on coniferous forest landscapes of the northern Rocky Mountains. USDA Forest Service Research Paper INT-RP-484. Intermountain Research Station, Ogden, Utah.

Keith, J.O. 1965. The Abert squirrel and its dependence on ponderosa pine. *Ecology* 46:150-63.

Lang, D.M., Stewart, S.S. 1910. Reconnaissance of the Kaibab National Forest. Northern Arizona University, Flagstaff, Arizona, USA. Unpublished report on file.

Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangelands: a viewpoint. *Journal of Range Management* 44:427-433.

Lindenmuth Jr., A.W. 1962. Effects on fuels and trees of a large intentional burn in ponderosa pine. *Journal of Forestry* 60:804-810.

Linhart, Yan B. 1988. Ecological and evolutionary studies of ponderosa pine in the Rocky Mountains. *In*: Baumgartner, David M.; Lotan, James E., compilers. Ponderosa pine: The species and its management: Symposium proceedings; 1987 September 29 - October 1; Spokane, WA. Pullman, WA: Washington State University, Cooperative Extension: 77-89.

Martinson, E.J., and Omi, P.N. 2003. Performance of fuel treatments subjected to wildfires. USDA Forest Service Proceedings RMRS-P-29.

McKenzie, D., Gedalof, Z., Peterson, D.L., Mote, P. 2004. Climatic change, wildfire, and conservation. *Conservation Biology* 18(4):890-902.

Miller, J.D., and Yool, S.R. 2002. Mapping forest post-fire canopy consumption in several overstory types using multi-temporal Landsat TM and ETM data. *Remote Sensing of Environment* 82:481-496.

Milner, K.S., Coble, D.W., McMahan, A.J., Smith, E.J. 2003. FVSBGC: a hybrid of the physiological model STAND-BGC and the forest vegetation simulator. *Canadian Journal of Forest Research* 33(3):466-479.

Moir, W.H. and Dieterich, W.H. 1988. Old-growth ponderosa pine from succession in pine-bunchgrass forests in Arizona and New Mexico. *Natural Areas Journal* 8(1):17-24.

Moore, M.M., Huffman, D.W., Fulé, P.Z., Covington, W. W., and Crouse, J.E. 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. *Forest Science* 50(2):162-176.

National Park Service/US Geological Survey Burn Severity Mapping Project. 2002. DNBR (differenced normalized burn ratio) map of the Rodeo-Chediski burn area. http://edc2.usgs.gov/fsp/severity/fire_main.asp

Omi, P.N., and Kalabokidis, K.D. 1991. Fire damage on extensively versus intensively managed forest stands within the North Fork Fire, 1988. *Northwest Science* 65:149-157.

Omi, P.N., and Martinson, E.J. 2002. Effect of fuels treatment on wildfire severity. Final Report. Submitted to the Joint Fire Science Program Governing Board. Ft. Collins, CO: Western Forest Fire Research Center, Colorado State University. Available online at: www.cnr.colostate.edu/frws/research/westfire/FinalReport.pdf

Pearson, G.A. 1923. Natural reproduction of western yellow pine in the Southwest. Bulletin 1105. Washington, D.C: USDA Forest Service.

Petraitis, P.S., and Latham, R.E. 1999. The importance of scale in testing the origins of alternative community states. *Ecology* 80(2):429-442.

Pyne, S.J. 1982. Fire in America: a cultural history of wildland and rural fire. Princeton, N.J.: Princeton University Press.

Pollet, J. and Omi, P.N. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. *International Journal of Wildland Fire* 11:1-10.

Retana, J; Espelta, JM; Habrouk, A; Ordonez, JL; de Sola-Morales, F. 2002. Regeneration patterns of three Mediterranean pines and forest changes after a large wildfire in northeastern Spain. *Ecoscience* 9(1):89-97.

Running, S.W., Coughlan, J.C. 1988. A general model of forest ecosystem processes for regional applications. I. Hydrologic balance, canopy gas exchange and primary production processes. *Ecological Modeling* 42:125-154.

Running, S.W., Gower, S.T. 1991. FOREST-BGC, a general model of forest ecosystem processes for regional applications. II. Dynamic carbon allocation and nitrogen budgets. *Tree Physiology* 9:147-160.

Sackett, S.S. 1980. Woody fuel particle size and specific gravity of southwestern tree species. USDA Forest Service Research Note RM-389. Ft. Collins, CO: Rocky Mountain Forest and Range Experiment Station. 10 pp.

Sackett, S.S., Haase, S.M. 1998. Two case histories for using prescribed fire to restore ponderosa pine ecosystems in northern Arizona. Pages 380-289 in: Pruden, T.L. and L.A. Brennan (eds.), *Fire in ecosystem management: shifting the paradigm from suppression to prescription*. Tall Timbers Fire Ecology Conference Proceedings, No. 20. Tallahassee, FL: Tall Timbers Research Station.

Savage, M., Brown, P.M., Feddema, J. 1996. The role of climate in a pine forest regeneration pulse in the southwestern United States. *Ecoscience* 3(3):310-318.

Savage, M. and Mast, J.N. 2005. How resilient are southwestern ponderosa pine forests after crown fire? *Canadian Journal of Forest Research*: in press.

Scott, J.H. 1998. Fuel reduction in residential and scenic forests: A comparison of three treatments in a western Montana ponderosa pine stand. Research Paper RMRS-RP-5. Ogden, UT: USDA Forest Service, Rocky Mountain Rsearch Station.

Scurlock, D. and Finch, D.M. 1997. A historical review. Pp. 43-68 in *Songbird ecology in southwestern ponderosa pine forests: a literature review,* ed. W.M. Block and D.M. Finch. General technical report RM-292. Fort Collins, CO: USDA Forest Service.

Stephens, S.L. 1998. Evaluation of the effects of silvicultural and fuels treatment on potential fire behavior in Sierra Nevada mixed-conifer forests. *Forest Ecology and Management* 105:21-35.

Stokes, M.A., Smiley, T.L. 1968. An introduction to tree-ring dating. University of Chicago Press, Chicago.

Swetnam, T.W., and Betancourt, J.L. 1990. Fire-Southern Oscillation relations in the southwestern United States. *Science* 249(4972):1017-1020.

Swetnam, T. W., and C. H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. In: Proceedings of the Second La Mesa Fire Symposium, 29–30 March 1994, Los Alamos, New Mexico. C. D. Allen, editor. USDA Forest Service General Technical Report RM-GTR-286. Pp. 11–32.

Swetnam, T.W., Allen, C.D., Betancourt, J.L. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9(4):1189-1206.

Tiedemann, A.R., Klemmedson, J.O., and Bull, E.L. 2000. Solutions of forest health problems with prescribed fire: are forest productivity and wildlife at risk? *Forest Ecology and Management* 127:1-18.

Thomas, J.W. 1979. Wildlife habitats in managed forests in the Blue Mountains of Oregon and Washington. U.S. Department of Agriculture, Forest Service, Washington, D.C. Agriculture Handbook No. 553.

USDA Forest Service. 1987. Forest and woodland habitat types (plant associations) of northern New Mexico and northern Arizona. 2nd edition. Albuquerque, N.M.: USDA Forest Service, Southwestern Region.

USDA Forest Service. 1999. Protecting people and sustaining resources in fire-adapted ecosystems – a cohesive strategy. First final draft. The Forest Service management response to General Accounting Office Report GAO/RCED-99-65. Washington, D.C.: USDA Forest Service.

USDA Forest Service. 2002. Rodeo-Chediski fire effects summary report: Apache-Sitgreaves National Forests. Washington, D.C.: USDA Forest Service. 74 pp.

USDA/USDI. 2004. Healthy Forests Progress Report, December 2004. http://www.healthyforests.gov.

U.S. Senate and House of Representatives. 2003. H.R. 1904: Healthy Forests Restoration Act of 2003. http://thomas.loc.gov.

Vandendriesche, D. 1994. CFI Field Manual: Phoenix Area Reservations. First Edition. U.S. Department of the Interior, Bureau of Indian Affairs.

Van Mantgem, P., Schwartz, M., and Keifer, M. 2001. Monitoring fire effects for managed burns and wildfires: coming to terms with pseudoreplication. *Natural Areas Journal* 21(1):266-273.

van Wagtendonk, J.W., 1996. Use of a deterministic fire growth model to test fuel treatments. Sierra Nevada Ecosystem Project, Final Report to Congress, Vol. II. Assessments and Scientific Basis for Management Options. Centers for Water and Wildland Resources, University of California, Davis, CA, pp. 1155–1166.

van Wagtendonk, J.W., Root, R.R., and Key, C.H. 2004. Comparison of AVIRIS and Landsat ETM+ detection capabilities for burn severity. *Remote Sensing of Environment* 92:397-408.

Vihanek, R.E., and Ottmar, R.D. 1993. When logged units burn in a wildfire, does slash treatment mitigate effects? In '12th conference on fire and forest meteorology, 26-28 October 1993', Jekyll Island, Georgia. Pp. 709-714.

Viro, P.J. 1974. Effects of forest fire on soil. In: Kozlowski, T.T., Ahlgren, C.E. (eds.), Fire in ecosystems. Academic Press, New York. Pp. 7-45.

Wagle, R.F., and Eakle, T.W. 1979. A controlled burn reduces the impact of a subsequent wildfire in a Ponderosa pine vegetation type. *Forest Science* 25(1):123-129.

Waltz, A.E.M., Fulé, P.Z., Covington, W.W., and Moore, M.M. 2003. Diversity in ponderosa pine forest structure following ecological restoration treatments. *Forest Science* 49(6):885-900.

Weatherspoon, C.P., and Skinner, C.N. 1995. An assessment of factors associated with damage to tree crowns from the 1987 wildfires in northern California. *Forest Science* 41(3):430-451.

Weaver, H. 1951. Fire as an ecological factor in the southwestern ponderosa pine forests. *Journal of Forestry* 49:93-98.

Weaver, H. 1955. Fire as an enemy, friend, and tool in forest management. *Journal of Forestry* 53:499-504.

Weaver, H. 1961. Fire as an ecological and silvicultural factor in the ponderosa pine region of the Pacific slope. *Journal of Forestry* 41:7-14.

Woolsey, T.S., Jr. 1911. Western yellow pine in Arizona and New Mexico. Bulletin no. 101. Washington, D.C.: USDA Forest Service.

_______. 1912. Permanent sample plots. *Forest Quarterly* 10:38-44.

Youtz, J. 2003. Vegetation management: silvicultural decision and prescription guidelines. U.S. Department of the Interior, Bureau of Indian Affairs, Fort Apache Agency, Branch of Forestry.

7. APPENDIX

As mentioned in the Discussion of Chapter 4, we did not attempt to reconstruct pre-fire forest structure for the study on White Mountain Apache Tribal lands because we did not have a full sample of dead trees. I include data for the dead trees we measured in this appendix, with the caveat that since we did not have a full sample, these results are an underestimate.

I also include data on canopy cover for both the White Mountain Apache Tribal lands and Apache-Sitgreaves National Forest studies. These data were produced using Gap Light Analyzer (Institute of Ecosystem Studies 1999) on hemispherical photographs taken with a 180º fisheye lens at each plot center.

Figure 7.1. WMAT post-wildfire forest density, including dead trees. Error bars represent ±1 stderr for this and subsequent figures.

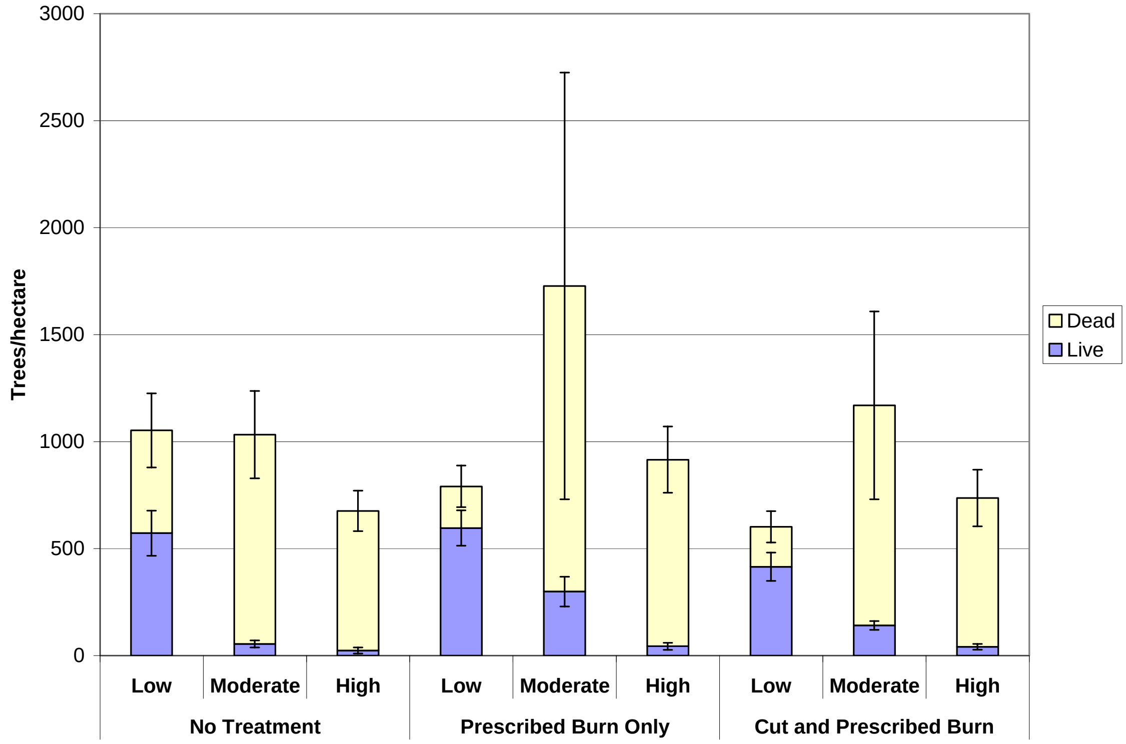


Figure 7.2. WMAT post-wildfire basal area, including dead trees.

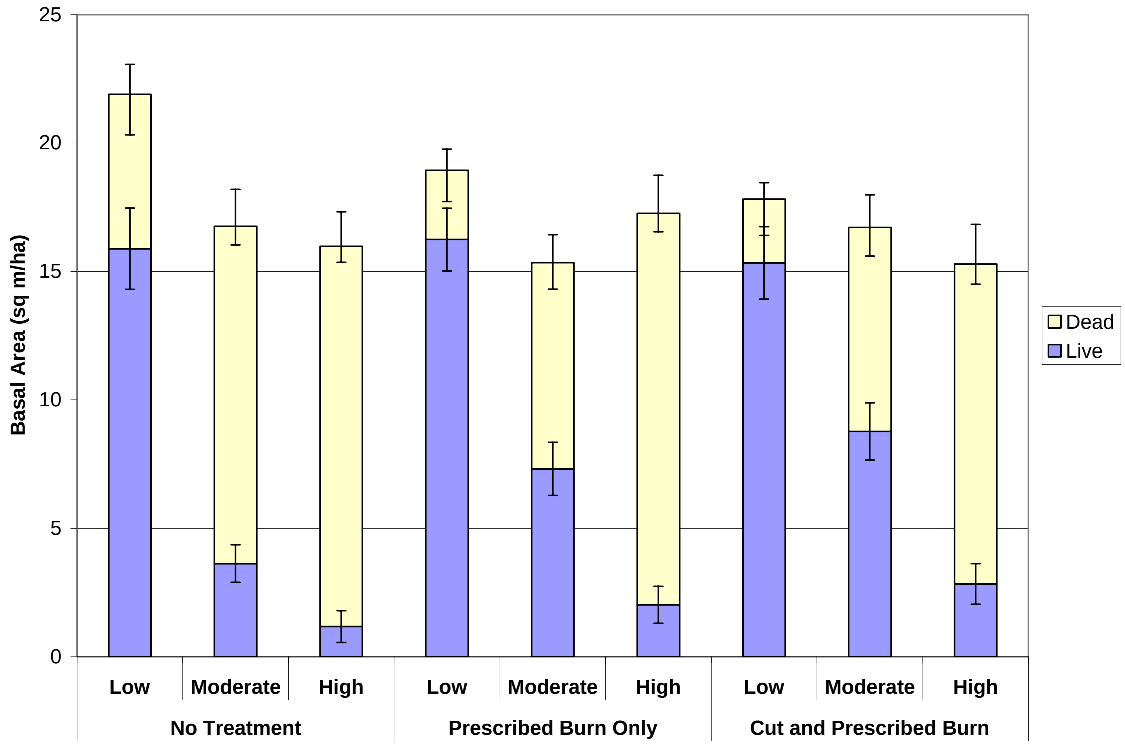


Figure 7.3. Post-wildfire diameter distribution of live ponderosa pine trees on WMAT.

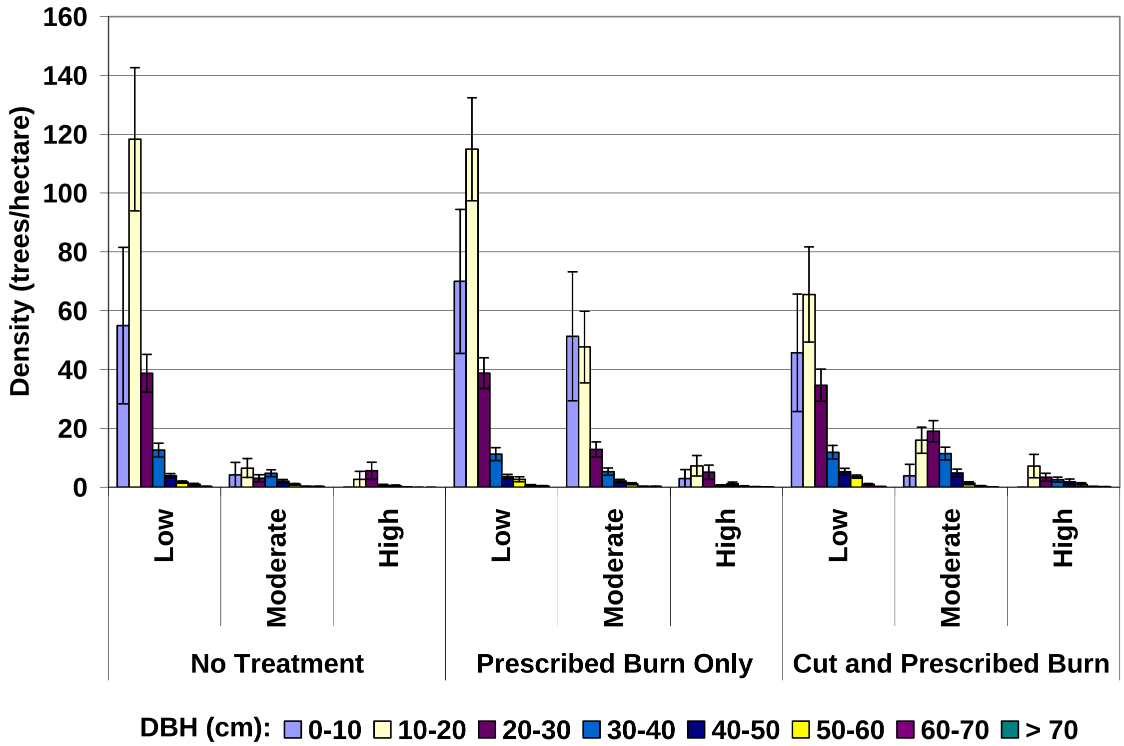


Figure 7.4. Post-wildfire diameter distribution of dead ponderosa pine trees on WMAT. Please note the difference in the amplitude of the y-axis (trees/hectare) compared to the diameter distribution for live trees (Figure 7.3).

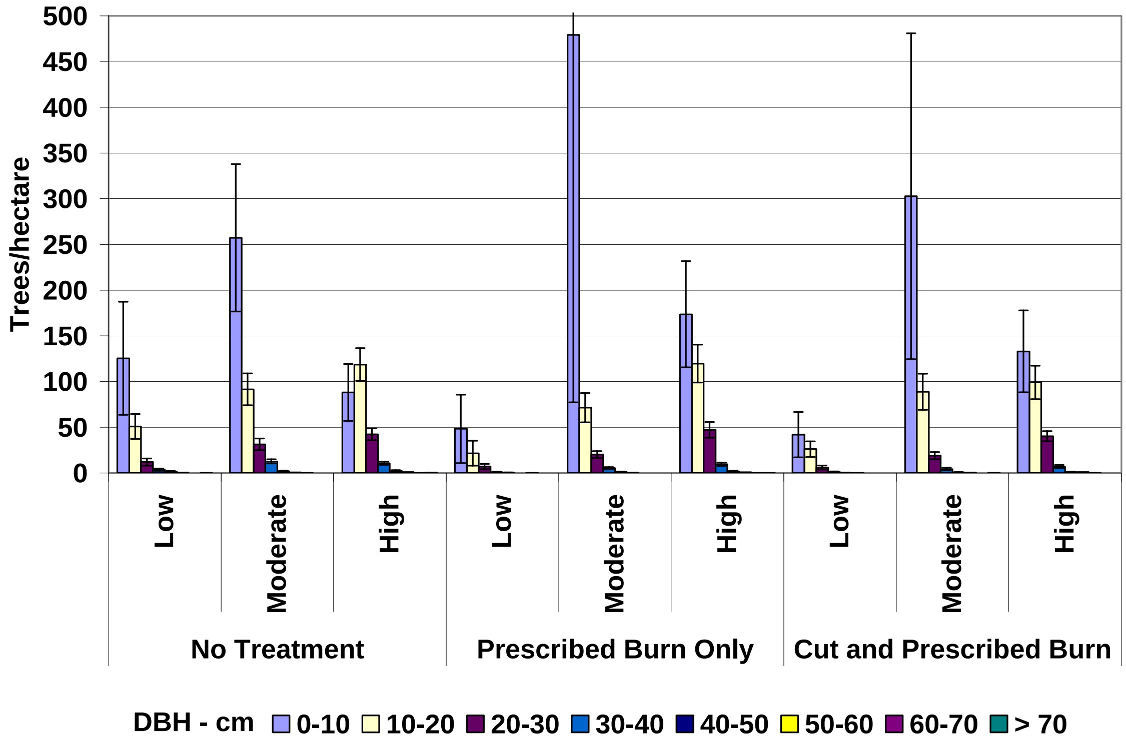


Figure 7.5. Post-wildfire canopy cover on WMAT, including standing dead trees.

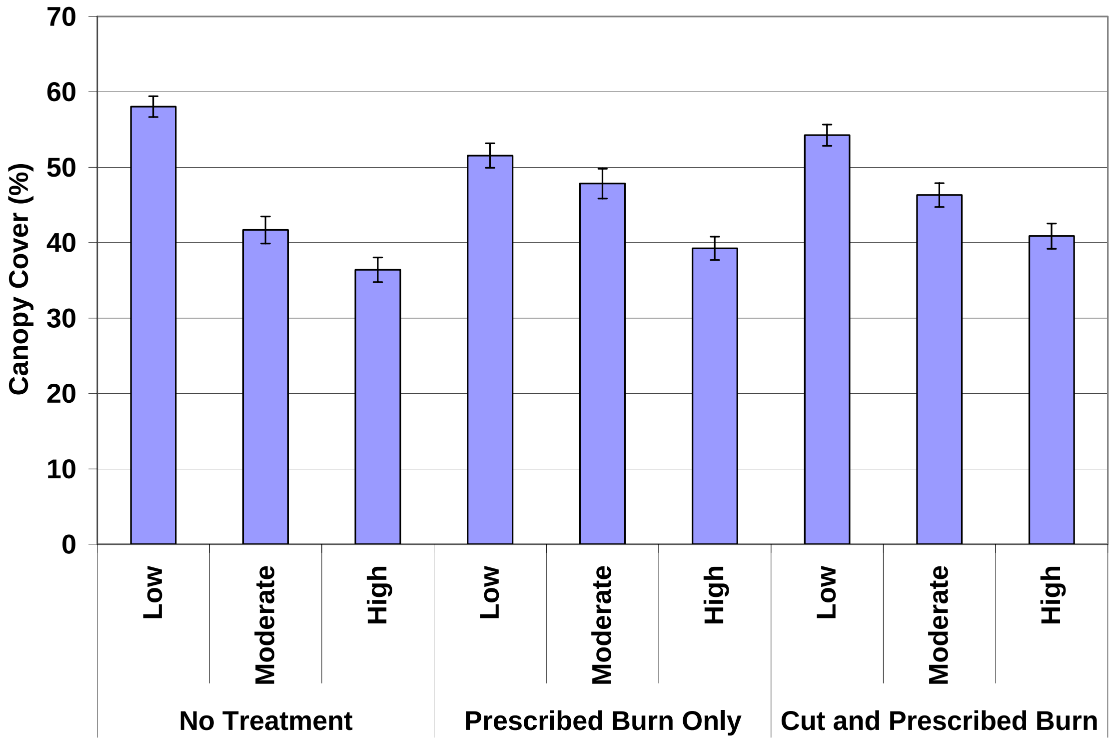


Figure 7.6. Post-wildfire canopy cover on the Apache-Sitgreaves National Forest, including standing dead trees.